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A RECIRCULATING WASTE SYSTEM FOR SWINE UNITS

J. R. Miner

Iowa State University

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A RECIRCULATING WASTE SYSTEM FOR SWINE UNITS

By

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The purpose of this project was to develop and characterize a swine manure management system. The goal of the system was to collect, transport, treat, reuse and dispose of the manure in such a way that it would be compatible with current confinement swine production systems, yet minimize both labor and pollution potential. Such a system was devised and evaluated. Its basis was to hydraulically flush manure from shallow dunging gutters with the treated wastewater. The treatment devices evaluated included an anaerobic lagoon and an oxidation ditch. Excess water from the system was applied under controlled observation to adjacent cropland using conventional sprinkler irrigation equipment. The overall validity of this concept was proven.

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FOREWORD

Man and his environment must be protected from the adverse effects of pesticides, radiation, noise and other forms of pollution, and the unwise management of solid waste. Efforts to protect the environment require a focus that recognizes the interplay between the components of our physical environment--air, water, and land. The National Environmental Research Centers provide this multidisciplinary focus through programs engaged in

- studies on the effects of environmental contaminants on man and the biosphere, and

- a search for ways to prevent contamination and to recycle valuable resources.

In an attempt to solve the problem of solid waste treatment, this study examines a waste system for swine units. This report, published by the National Environmental Research Center - Cincinnati, devised and evaluated a swine manure management system that would be compatible with current swine production systems.

A. W. Breidenbach, Ph.D.
Director
National Environmental
Research Center - Cincinnati

ABSTRACT

The purpose of this project was to develop and characterize a swine manure management system. The goal of the system was to collect, transport, treat, reuse and dispose of the manure in such a way that it would be compatible with current confinement swine production systems, yet minimize both labor and pollution potential. Such a system was devised and evaluated. Its basis was to hydraulically flush manure from shallow dunging gutters with the treated wastewater. The treatment devices evaluated included an anaerobic lagoon and an oxidation ditch. Excess water from the system was applied under controlled observation to adjacent cropland using conventional sprinkler irrigation equipment.

The overall validity of this concept was proven. The system eliminated the need for manual manure handling within the building, improved pen and animal cleanliness, reduced odors, and eliminated manure hauling or stream discharge all without adversely affecting the performance of animals within the building. Operating problems were largely related to equipment failures, caused mostly by the necessity to use adapted equipment designed for some other intended operation or fabricated specialty items not otherwise available.

The report includes complete operating data on the waste management system and a record of the various equipment problems and their alleviation. The anaerobic lagoon effluent was found safe for recirculation and represented least cost alternative. Using the lagoon and oxidation ditch in series produced on improved effluent quality but at a somewhat increased cost. Using the oxidation ditch only resulted in solids handling problems and further increased costs.

This report was submitted in fulfillment of Grant No. RO 1 EP 00283 from the Bureau of Solid Waste Management, Public Health Service, Department of Health, Education and Welfare.

Key words: Swine wastes, reuse, oxidation ditch, lagoon, B.O.D. removal, C.O.D., solids reduction, waste treatment, irrigation disposal.

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SUMMARY AND CONCLUSIONS

Accomplishments

The work described in this report was performed with the objective of determining the practicality of a recycled effluent system for manure transport in swine finishing buildings. The system was developed around Unit K, an existing 700-head confinement swine-finishing building erected in 1960. Manure transport, treatment methods used, recycled effluent quality, fluid handling equipment as well as treatment and disposal of the excess from the system are evaluated.

This project resulted in the development of a swine-finishing facility in which 700 animals are confined on a continuous basis, in which manual labor for manure removal has been almost totally eliminated, and from which there is no waste discharge to surface streams. The building is less odorous than conventional swine production units because of the frequent manure removal. The only net waste product is excess liquid which is applied to cropland during the summer using conventional irrigation equipment.

Based upon the success of this project a demonstration project supported by the Environmental Protection Agency was instituted. The demonstration project includes eight buildings fitted with flushing gutters and incorporating three independent waste treatment systems all of which produce an effluent to be used in the flushing system. This facility is to be placed in operation during the summer of 1971. In addition to this demonstration project, the concepts developed in this research have been incorporated by numerous midwestern swine producers.

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Manure Handling

In the early days of Unit K, the use of a small continuous stream of fresh water through the dunging gutters indicated that hydraulic manure transport was a promising manure removal system. More recent experience suggests that, with flushing tanks, 4 gpd per pig discharged in regular 2 hr interval flushings will transport the manure adequately in this 120 ft long building containing 350 animals to each gutter. Pen cleanliness was improved even more by reducing the interval between flushes but this also increased the volume of manure slurry requiring treatment. The per pig value of transport liquid given is only approximate as no tests were performed to find an optimum relation between total flow quantities and individual pig quantities. Moreover, all the work described was performed using a fixed channel slope of 1 ft in 120 ft. Gutter width and depth are other variables not examined in detail. Visual examination of slurry flow pattern just downstream of the tank, and at the far end of the gutter, does suggest that liquid velocity in the bulk of the channel is more a function of the channel geometry than the rate of discharge at the head. It also appears that the minimum total quantity of liquid required to satisfy the hydraulic capacity of the gutter is independent of the number of pens along the gutter. It was found that an unoccupied 42 in. gutter width was too great because manure particles may settle out and the slurry then takes a meandering course between banks of settled manure. In the test facility this meandering could be overcome by reducing the channel width to 15 in.

Phase 1. Anaerobic Lagoon and Oxidation Ditch in The System. 1969

Operation was started in February 1969 and continued until May. The flow process was to hydraulically transport wastes from a 700 head swine finishing building (Unit K) to initial treatment in an anaerobic lagoon, then to an oxidation ditch and a final settling tank whereupon the clarified effluent from the settling tank was returned for manure transport in Unit K. Derived fecal parameters, reduced to a unit pig basis, compared favorably with values reported in the literature.

The anaerobic lagoon functioned as a useful part of the system. It reduced VS, BOD and COD by more than 85%, though N was only reduced by 60%.

The lagoon also served to reduce coliforms by 99%. No evidence of a trend in the concentration of dissolved inorganic solids could be determined in the 4 mo recycling period. Vigorous biological activity, as evidenced by rising boils of gas and bottom material, occurred whenever the lagoon temperature was above 55°F. Some possible machanisms of nitrogen reduction are discussed, including precipitation as Mg and Ca ammonium phosphates and dilution effects. In the Spring, biological resurgence brought an unexpected increase in phosphate levels, attributed to two sources in the bottom sediments; one supply of P was organic matter but the other, most probably, was inorganic P in the form of Mg and Ca phosphates. Deposition of $Mg(NH_4)PO_4$ on equipment submerged in the lagoon is cited as contributory evidence.

The clarified effluent from the oxidation ditch, though slightly yellow and turbid, was odor-free and contained a BOD_5 of less than 120 mg/l. Food to microorganism ratios, examined throughout the test, were all lower than 0.05 lb BOD_5/lb MLSS. Until the upsurge of lagoon activity caused a reduction in influent strength to the ditch, the rate of production of volatile suspended solids, as derived by linear regression, was 16.2 lb/day. A cell yield parameter could not be obtained because of other uncertainties. The generation of VSS was examined in relation to phosphate uptake and the results indicate that the sludge was storing 7% P (dry weight basis). The excess aeration capacity of the rotor caused extensive nitrification after the ditch temperature climbed above 39°F. Power requirements, per unit of BOD_5 removed, were higher than values normally reported in the literature because of the excess aeration capacity. The power requirement per pig was 0.09 kwh/day.

Phase 2. Manure Diverted to The Ditch. 1970/71

This research began in August 1970 and ended in March 1971, with a major breakdown during October and November. The raw manure slurry from Unit K flowed directly to the oxidation ditch, the settled effluent from the ditch then returned to Unit K for transporting manure. The anaerobic lagoon served only as a receptor for the overflow from the ditch.

Reductions in VS, COD and N obtained in the ditch were all in excess of 75%. Nitrogen reduction now averaged 80% and the contribution of nitrification and denitrification is discussed. Food to microorganism ratios were calculated as about 0.05 lb BOD_5/lb MLSS during most of the operation. Lack of aeration capacity, hair and husks were all considered as factors contributing to the poor settling observed after the MLSS exceeded 12,000 mg/l. The small total volume of the ditch allowed detection of the expected trend of dissolved inorganic salt concentration. Experimental results correlated well with a theoretical exponential equation. Actual rates of accretion of Cl were compared with the literature, and inhibition from salt toxicity was rejected as a concern for this or a comparable system. Power requirements were much higher than in Phase 1, though the power consumption per unit of BOD_5 (0.705 kwh/lb BOD_5) was now more comparable with municipal experience. The power requirements for the individual pig increased to 0.24 kwh/day.

Aerobic Sludge Digestion. 1971

After ending Phase 2 the liquor remaining in the ditch was allowed to digest aerobically, without further feeding, for 10 wk. After the third week, settling improved markedly. For convenience in examining the results, an exponential decay model was selected, with the exponents being determined for volatile matter, COD and Kjeldahl nitrogen in the suspended solids. A sudden rise in all parameters, occurring after 30 days, reduced the agreement between the model and the results. This rise was most probably due to some sediment being resuspended. Comparison of the values of the exponents, and also the ratios VSS:COD:N, showed that the N content of the sludge decreased most rapidly. Aerobic sludge digestion was judged beneficial with regard to VS reduction and improved settling.

Liquid Handling

The poor settling, hair and husk laden ditch liquor obtained in Phase 2 presented problems. Whereas the low solids content ditch liquor in Phase 1 had settled readily and the supernatant liquor was easily pumped, returning pumping of settled sludge back to the oxidation ditch finally required the use of a large bore diaphragm pump. Based on these experiences, aerobic treatment of raw manure at MLSS above 10,000 mg/l cannot be recommended if the liquor is to be settled and reused. A larger ditch volume and controlled sludge discharge could overcome the encountered problems but they would likely introduce equally serious management problems for most producers. Pumping anaerobic lagoon supernatant did produce some problems because of the deposition of $Mg(NH_4)PO_4$. It is unlikely that this can be overcome completely, but difficulties can be minimized by using plumbing with few fittings, by selecting suitable pumps, and by occasional flushing of the system with an appropriate solvent. Experience has shown that flexible impellor pumps work well. Flushing tanks using mechanical valves were useful for research purposes, but a better design uses a syphon having no moving parts. The design of low cost, lightweight syphon tanks has been described by Person and Miner (57).

Pigs' Water Supply Phase 2 Treatment System. 1971

An experiment was conducted to determine the effect on the pigs of being forced to drink out of the flushing gutter, no other drinking water being available. Statistical analyses of weight gain, feed consumed and feed conversion were performed. Both weight gain and feed conversion were shown to deteriorate (5% level). TGE virus was monitored as it passed through the treatment system. No diminution of viral intensity, in ditch or lagoon, was noted until the water temperature exceeded 60°F.

Irrigation Disposal 1968

Irrigation was begun in June 1968 and continued until the following November. The irrigation equipment worked well without noticeable corrosion or plugging from suspended materials. Odor during sprinkling was not a problem or was insect breeding encouraged since no permanent ponding occured. Infiltration rates remained approximately constant throughout the season. Four treatments were applied, 1.5 and 3 in. irrigation quantities at 70% and 90% available moisture levels. Application rate was 0.30 iph and, although less than recommended for clear water, this was found the maximum feasible rate using lagoon liquor. Samples were collected from tile drains installed at 48 in. and at intermediate depths from buried pan lysimeters. Differences between treatments were small. An average of 97% of the applied COD was removed by the time the effluent reached the tile, 50% removed in the top 3 in. of soil. Nitrogen was removed as the effluent moved through the soil and it was concluded that biological denitrification was the predominant phenomena causing the 80% reduction observed. Phosphorus removal was judged excellent since percolation removed 99% with 83% being removed in the top 3 in. of soil.

Irrigation Disposal 1969-70

Irrigation was practiced from May to October during both seasons. The plots continued to accept water readily but problems with cracking of the soil in the tile field trenches caused some short circuiting from the surface to the tile.

Porous cup samplers and soil moisture tensiometers were installed and these allowed more information to be obtained over a greater range of soil moisture content. Treatments in 1969 were 2 in. of clear water, water spiked with a known amount of nitrogenous fertilizer and lagoon water. These three treatments were applied at 750 mb tension and a fourth treatment of lagoon water was also applied at 350 mb. In 1970, only four applications of lagoon water were used; in one amount of 2 in. and 3 daily amounts of 2/3 in. The two moisture tensions were 750 mb and 350 mb. Fertilized water behaved as lagoon water with regard to nitrogen removal. The 1970 season showed that small consecutive applications were better than one major application if transient tile effluent strengths were of major importance, though removals on a mass basis were nearly the same for both regimes. Mass removal of COD ranged from 79 to 93%. Better removal was obtained with the 3 daily 2/3 in. applications. COD removal deteriorated as the irrigation season progressed. Porous cup sampling indicated 89% removal of COD at 6 in. Nitrogen removal allowing for various water losses, amounted to a mass removal of 49 to 63%. This was inferior to the 1968 result of 86% but the soil cracking introduced short circuiting of NH_3-N.

Useful decreases in N were obtained between the sprinkler nozzle and the ground due to ammonia desorption. Denitrification was examined in the laboratory and lagoon water organic carbon was judged a poor substrate, solutions of nitrate in lagoon water denitrified most rapidly if dosed with glucose. Phosphorus removal remained high (90 - 97%) but some indication of a linear relation between accumulated applied P and tile drainage became apparent. In three years the chloride ion removal had declined to zero indicating dynamic equilibrium with the soil solution.

Desorption of Ammonia From the Anaerobic Lagoon

Loss of nitrogen from the anaerobic lagoon over an annual cycle was well established, but could not be entirely explained by sedimentation and precipitation. Ammonia can migrate across the liquid-vapor boundary of water, the direction and rate of migration dependent upon such factors as partial pressure differentials, pH, temperature and relative motion between liquid and vapor. Examination of the literature revealed that the experimental results obtained in the field could be made to yield kinetic rate coefficients for desorption that compare quite favorably with published absorption results. It was concluded that between May and October ammonia desorption could account for 80 to 145% of the N influent to the lagoon.

OBSERVATIONS

Manure is composed of the same basic building blocks as food: protein, carbohydrates, vitamins and minerals. Future waste handling systems will be designed to recover these in a useful form.

A typical Midwestern swine manure slurry, examined physically, breaks down into three basic fractions; a supernatant containing soluble and colloid matter, a fine settleable particulate fraction and a course settleable particulate fraction. There is mounting justification that systems which segragate these fractions will be superior to those that handle them together. The fractions should separate readily because they have physically different properties; but little work has been done to examine the chemical properties of each fraction in detail, a necessary step before firm recommendations for their treatment can be made.

A reasonable insight into the probable processes can be developed, however. The coarse, fast settling fraction consists of undigested corn particles, lignaceous husk material and hair. Such material should be re-fed on an experimental basis to find out if a better total fraction of the corn may be used, or if excess addition of bulk material, such as husk and hair, simply lowers animal performamce. This coarse material could probably be drained so that it would not putrefy in storage and landspread at a convenient time. Alternatively a large livestock producer might examine the use of this fraction as fuel to power an engine and generator.

The fine particulate fraction is largely unquantified. However, it appears to be metabolic by-products that will surely be very putrescible. Of the many possibilities that present themselves, two seem worth examining. Since the material is finely divided, quite simple chemical treatment might render it suitable for re-feeding. Another alternative might be incineration and use of the ash as fertilizer. The ash would, at least, concentrate P and K to a bulk comparable with commercial fertilizers. Nitrogen loss to the atmosphere does need to be controlled to avoid production of NH_3 and NO_X as air pollutants.

The liquid portion remaining after fractionation putrefies readily and will need immediate treatment. This liquid free of large particulate matter, will therefore treat easily using any of the established biological processes. The remaining COD after fractionating will be much lower than that of the manure slurry, hence, treatment in an aerator may be economically attractive. Two alternative aerobic processes which need very little power, compared to direct aeration, are the Rotating Biological Contractor and the high rate trickling filter using plastic media. In any case, facilities will need to be provided for excess sludge treatment and for excess liquid disposal.

Additional work is also indicated to refine the flushing gutter design to improve manure transport. The 42 in. wide gutter used in this facility proved satisfactory when pigs had access to the area but was not self-cleaning when pens were empty. Reduction of the gutter width is helpful in this respect but such reduction is limited by animal acceptance and training habits.

INTRODUCTION

Renovation and Reuse of Wastewater

Man is now past the point from which he could look at water as a disposable and unlimited commodity. Before the industrial revolution, cities were quite small and could satisfy their water needs from rivers without worrying about, or even comprehending, pathogens or pollution. But by the mid-19th Century, the rural migration to the cities had caused an explosion in city populations and untreated water was no longer potable without considerable risk of infection. Fortunately, the impetus given to science by the industrial revolution had revealed the knowledge necessary to institute treatment facilities for both drinking water and for wastewater. So, by the beginning of this century the spectre of massive epidemics of cholera, typhoid and other water-borne diseases had been laid to rest.

However, man is now entering an era of water shortage; because his phenomenal success at organizing an industrial society has led him to expect extensive material possessions as the norm. Man's eating habits have also change. As he becomes more affluent, he eats more animal than vegetable protein and his consumption of animal protein is increasing (79). This increasing appetite of man, the consumer, means that the number of units of industrial output required to keep an individual fed, watered, clothed and amused increases daily. Essentially all these industrial processes use water as a raw material. However, this is not the whole story. Man's water use is rising annually, and so is the number of men.

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Although there are certain industrial processes, such as condenser cooling, that leave water unchanged chemically, most other industrial and agricultural processes cause an increase in the burden of dissolved and suspended matter carried by the water. The water and sewage treatment systems developed in the last part of the 19th Century were quite adequate for removal of organic matter, but they left the concentration of inorganic materials largely unchanged. The end result of this treatment deficiency was to render the water more saline after each use. At first this was not significant, since rainfall was sufficient to dilute the salts before the next user took in the water.

Today there are only two low cost sources of new water; surface storage in lakes and rivers, and groundwater storage in aquifers. Berger (9) reports that the United States has only a limited supply of water available from these sources and that the country's requirements will exceed this by 1980. Berger (9) also notes that conventional secondary treatment, even if applied to all wastewaters, will not suffice; because the natural purification cycle does not adequately cope with such materials as refractory industrial organic compounds, viruses, and certain pathogens such as Salmonella. To some degree these materials pass through conventional treatment systems, making the task of treating the water for drinking increasingly more difficult.

Water resources are not evenly distributed throughout the United States; hence some states have encountered supply problems well before the 1980 deadline predicted by Berger (9). Two such states, Texas and California, have begun practicing water reclamation. Fleming and Jobes (20) state that by 2020, Texas is expected to require 28 million acre-ft of water for all purposes, double that used in 1960. Texas has an estimated safe yield of 21.3 million acre-ft of water from all sources; thus the state already has extensive experience with water reuse. The economy of Texas leans heavily upon oil and its derivatives, and these industries are heavy water users. However, Rickles (63) has shown that large integrated petrochemical complexes, such as that at Odessa, can make good use of reclaimed water. Foster and Jopling (22) have summarized California's attitude toward water reuse. At this time the state does not envisage direct reuse of effluent for potable supplies, but it is actively promoting reuse for irrigation and industrial cooling. California divides reclaimed water into three categories; primary effluent, oxidized wastewater and filtered wastewater. Some control over quality standards was obviously necessary, but the state did not wish to inhibit potential reusers by imposition of rigorous chemical standards. Instead, coliform counts were used since these might be performed with a minimum of skill and technical equipment, following development of some simplified techniques. As an example, the state code will allow spray irrigation if the water meets a most probable number of coliforms less than 2.2/100 ml.

Garthe and Gilbert (24) reviewed one of the country's longest running water reclamation projects. The Grand Canyon in Arizona has tourist accommodation on the South rim, but the major water bearing strata are on the North rim. Because potable water is at a premium on the South rim, water from an activated sludge plant has been reclaimed by filtration through anthracite followed by chlorination. This water has been reused for water closets, irrigation of gardens around the hotel and for the boilers. Reclamation has been practiced since 1926 without ill effects.

Not all authorities agree on the wisdom of using wastewater for agricultural irrigation. For example Hirsch (27) discusses the effects of metallic cations. The effect of sodium on soil is well documented and irrigation waters are characterized by their sodium absorption ratio,

$$\mathrm{SAR} = \frac{\mathrm{Na}}{\sqrt{\dfrac{\mathrm{Ca} + \mathrm{Mg}}{2}}} \tag{1}$$

Today's effluents may also contain boron, copper, chromium and other potentially toxic cations. Hirsch (27) presents a table showing that 0.9 - 1.4 mg/l of boron was found in the Rose Canyon reclaimed irrigation water. A boron concentration of 1 mg/l is generally accepted as safe for irrigation water.

Thus far, only simple systems such as primary settling, secondary biological treatment and chlorination have been discussed. Although these are often adequate for irrigation, industrial or potable reuse may require more advanced treatment. This tertiary treatment is required for removal of inorganic material, trace refractory organic material and viruses. Thus industrial reuse may entail chemical coagulation, lime treatment, ion exchange, activated carbon, etc. Koenig and Ford (39) have examined the various processes available and have presented some cost information. Evaporation will remove the major part of all contaminants, but it can cost $1.45 per 1000 gal at a flow rate of 10,000 gpd for a single effect evaporator. At the other end of the scale natural purification in an oxidation pond may be adequate, and this will only cost $0.011 per 1000 gal at 10,000 gpd. Koenig and Ford (39) point out succinctly that the economics of reuse will depend upon the following parameters:

Cost of raw water
Quality of raw water
Cost of treatment for disposal
Cost benefit of product recovery from the waste stream

As an example they mention a case in which 150 lb/day of Cr^{+6} was being discharged to a sewer. Not only did this amount to a $100/day loss to the plating firm but also Cr^{+6} is biologically toxic and unwelcome at municipal sewage treatment plants. Equipment was installed to recover the chromium and this recovery paid for the equipment.

Industrial areas which have arid climates have all encountered water resource deficiencies (20, 22) but only one country appears to have progressed as far as water reuse for municipal drinking supplies. Stander and VanVuuren (72) have described the research necessary to develop a 1.2 mgd demonstration plant at Windhoek, the capital of South West Africa. The processes employed are of some interest since they indicate the degree of complexity required to produce potable water. Water is taken from an oxidation pond fed with secondary effluent. This water is dosed with lime and passed through a flotation process, removing algae and some fraction of the phosphorus and nitrogen. The high pH of the effluent from the liming process is capitalized upon by following the flotation with ammonia stripping. Calcium and a large fraction of the suspended solids are then removed by chemical coagulation following carbonation. Sand filtration and foam fractionation, for detergent removal, are followed by activated carbon filtration and final chlorination. Stander and VanVuuren (72) present data showing that removal of phosphorus, organic matter, detergents, pathogens and viruses is essentially complete; but they do point out that nitrate nitrogen is little affected by the processes. Windhoek now has one-third of its water supply provided by this plant and public acceptance has been very good.

Lake Tahoe in California offered a rather different engineering challenge. Culp (18) discusses the treatment necessary to provide an effluent of sufficient quality that can be returned to the lake without danger of eutrophication. The system chosen is very similar to that outlined by Stander and VanVuuren (72) except that the effluent is taken from the final clarifiers of an activated sludge plant, thus obviating flotation to remove algae. All solid wastes from the processes are thermally processed to recover the lime and activated carbon.

To this point the literature reviewed shows that water may be reused for all purposes if sufficient treatment is provided. Even in the United States a large portion of future needs will have to be met from reclaimed water. Agriculture is a large water user and has practiced water reclamation unknowingly for some years. Irrigation and groundwater recharge just amount to using the soil as a treatment process. Although irrigation has accounted for the major fraction of agricultural water use, the changing pattern of livestock production is beginning to demand attention. Livestock in confinement present a new water demand because hydraulic transport promises low labor and equipment requirements. One of the pioneers in this field was Hercules whose labors are described by Apollodorus (5).

> The fifth labour he laid on him was to carry out the dung of the cattle of Augeas in a single day. Now Augeas was king of Elis; some say that he was a son of the Sun, others that he was a son of Poseidon, and others that he was a son of Phorbas; and he had many herds of cattle. Hercules accosted him, and without revealing the command of Eurystheus, said that he would carry out the dung in one day, if Augeas would give him the tithe of the cattle. Augeas was incredulous, but promised. Having taken Augeas' son to witness, Hercules made a breach in the foundation of the cattle-yard, and then, diverting the courses of the Alpheus and the Peneus, which flowed near each other, he turned them into the yard, having first made an outlet for the water through another opening.

Water quality standards in Greece must have been rather lax at that time.

Johnson (32) has reported a more practical system which was used for poultry. The birds were kept in cages over a rectangular section channel. The channel was connected to a three compartment settling tank. Effluent could be pumped from the last compartment to the head of the channel. This effluent was contained behind a moveable scraper placed in the channel. The hydraulic forces propelled the scraper along the channel shunting the accumulated manure out. Johnson (32) operated the system for four months, during which period dissolved solids built up appreciably. The settled solids in the tanks were field spread at convenient intervals. Witz _et al_. (86) used a similar system to Johnson's (32); however, they objected to the smell of the returned effluent. They were able to control odor by coagulation with alum, or by aeration in the settling tank. No details were given of the aeration device or its capacity. Pratt _et al_. (59) have described a pilot plant system for two beef animals. The animals were kept on a slotted floor, and facilities were provided for flushing the manure accumulation out from under the floor. The slurry first entered a 600 gal settling tank and then flowed into a 1000 gal treatment basin, effluent from the second tank was used for underfloor flushing. Three methods of treatment were applied to the 1000 gal tank; simple settling, aeration and aeration combined with chemical coagulation. It was found that none of these could provide a completely odor free effluent, although coagulation with aeration did provide a useful reduction in BOD_5. Clayton and Feng (15) used a pilot plant to treat the manure from one dairy cow. About 33 gal of settled effluent was added to the 7 gal of fresh manure from the cow, and this mixture was fed to a 2000 gal primary sedimentation tank. The effluent from this tank flowed into a 1000 gal aeration tank and was finally settled in a 200 gal sedimentation tank. Both sedimentation tanks were maintained at 70°F but the aerator was held at 95°F. No sludge return was made to the aerator. The system operated for 78 wk, producing an odorless effluent with a 90% reduction in BOD_5. The only major problem mentioned was a heavy scum buildup in the primary sedimentation unit. The scum, which was removed after 53 wk of operation, was described as being dense and recognizable as the fibre from bedding and silage. The material was easily handled with a fork and manure spreader, the odor was described as not being too disagreeable. Although the recycled effluent was stable and odorless, it was dark brown in color; and Clayton and Feng (15) doubt that the reuse of such an effluent in a dairy unit would be acceptable from the public health standpoint. The aeration capacity was generous since nitrification was established early, continuing at a high level, 300-600 mg/l, throughout the test. Foaming was not a problem in the aerator.

Smith and Jenkins (69) were concerned that increasing salt levels might prove inhibitory to treatment. Bench scale aeration units were established using poultry manure, but after one year the increase in salt concentrations was small. Hence Smith and Jenkins (69) had analyses performed on the raw manure to determine the inorganic ions present, and their relative magnitude. Solutions of these ions were then prepared in strengths of 0x, 1x, 5x, 10x and 100x. The 10x concentration approached that of normal sea water. When the aerobic digesters were established using these solutions, it was found that treatment efficiency was not significantly different among concentrations, except for the 100x concentration. Thus it was concluded that salt buildup should have no effects on biological treatment up to 20,000 mg/1, but that a concentration of 250,000 mg/1 does impair treatment.

Oxidation Ditches

The aerobic process of biological treatment will generally produce a lower strength of effluent than the anaerobic process. The two aerobic processes most commonly found in municipal pollution control plants are the trickling filter and the activated sludge process. The trickling filter has not seen extensive use for high strength wastes in the United States but it has received some attention in Britain (Eden et al. (19)), following the development of plastic media.

McLellan and Busch (51) have examined the fundamentals of activated sludge treatment. They have shown that as influent strength rises the mixed liquor suspended solids must also rise in order to produce low effluent strengths. This concept is very significant when applied to agricultural and industrial wastes, because the MLSS may now be as high as 30,000 mg/l, compared with 2000 mg/l for municipal wastes. These high solids levels are difficult to handle in conventional sparger aeration plants, hence recent years have seen a revival of interest in mechanical, sometimes called surface, aeration. Such aerators have more control over the partition of mixing vs. oxygen transfer.

One of the few innovations in sewage treatment was introduced by Passveer (56) in 1957. He used a shallow trench in the form of a closed loop. The aerator is a "brush" or "paddle wheel" with its axis parallel to the water surface, and normal to the center line of the trench. As first conceived, the oxidation ditch was merely a "canal with earthen slopes". This simplicity, and low construction cost per unit volume, has tended to persuade designers that the oxidation ditch is ideal for extended aeration activated sludge. At first it was thought that continuous return of solids from the final clarifier would drive the system into endogeneous respiration so that:

$$\frac{\text{New cell growth from}}{\text{substrate utilization}} \ \cdots$$

New cell growth from substrate utilization = Solids destruction by endogenous respiration

However, more recent work, typically that by Washington and Symons (80), shows that some biologically refractile material is continually being produced and this will accumulate in the system. Even if volatile solids destruction is not complete, the oxidation ditch, used as an extended aeration system, does offer the potential of large volatile solids reduction, a stable, well mineralized sludge and simplicity in operation.

The majority of oxidation ditches constructed up to the present have been for isolated rural communities or as quickly constructed additional facilities for an existing overloaded plant (Passveer (56), Burchinal and Jenkins (14)). Since strong industrial wastes are often best treated by extended aeration, the literature is beginning to contain reports on ditches which have been used for more exotic wastes. Adema (1) has described "the largest oxidation ditch in the world" for the treatment of coking and chemical plant wastes in Holland. This monster has a capacity of 30,000 m^3 ($1.06 \times 10^6 ft^3$) and is fitted with 10 rotors, each driven by a 80 hp motor. Some initial difficulties were experienced because the waste was below design strength. Excess aeration capacity soon promoted nitrification and, since the waste was insufficiently buffered, the pH dropped to 4.4, severely inhibiting further biological activity. This phenomena did not occur when the plant was more heavily loaded.

Scheltinga (67) has pioneered the use of oxidation ditches for the treatment of agricultural wastes in Holland. Total return of manure to pasture would lead to overfertilization with P and K which might adversely affect the animals grazed on it. This, and the growth of confinement livestock housing, has forced the country to handle its farm wastes by nontraditional methods. Discussing the application of oxidation ditches to calf and poultry waste, Scheltinga (67) concludes that BOD_5 loading per mass of solids carried is the only rational basis for design. He suggests 0.05 lb BOD_5 day/lb MLSS in the systems. Whereas 6000 mg/l would be a realistic MLSS concentration, 12,000 - 15,000 mg/l may be carried under some circumstances, though excess sludge production will not drop in proportion. Pontin and Baxter (58) have described a two ditch system for pigs. The first is an underfloor ditch and the effluent from this flows into a second ditch for polishing. In practice the first ditch served mainly to keep the manure mixed and to control odor. The MLSS in the second ditch rose to 20,000 mg/l after 12 wk because of the poor treatment effected in the first ditch. Effluent from the second ditch was passed through a final settling tank but the supernatant still contained a mean of 620 mg/l SS and a BOD_5 of 590 mg/l.

Jones et al. (34) have recently published the results of several years study on underfloor oxidation ditches. Initial studies using laboratory aeration vessels showed that a ditch volume of 8 ft^3 per finishing pig provided good treatment and no serious foaming. Although the MLSS of this unit was 21,000 mg/l, after 13 weeks the mean BOD_5 of the supernatant was only 45 mg/l. Although Jones et al. (34) are optimistic about oxidation ditches in swine confinement buildings, they do note that the clarified effluent will need further treatment before discharge. They suggest oxidation ponds. A recent study based on design parameters from Jones et al. (34) work has been reported by Windt et al. (85). Their underfloor ditch treated the waste from 330 pigs. The ditch was located on a commercial farm and marketing practices affected the loading rate. Approximately half the pigs were marketed at a time, hence there was a peak loading on the ditch at 6) day intervals. Evidence is presented that the MLSS showed a cyclic variation in accord with the variable loading. Ditch level was maintained constant by an overflow weir, thus solids were continually discharged. This overflow was metered so that hydraulic detention times could be established. Windt et al. (85) were surprised by the large overflows measured, much in excess of reports by other workers such as Jones _et al_. (34). The disparity was attributed to the pigs playing with their waterers. The ditch had a working volume of 3300 ft^3 and a designed hydraulic detention time of 50 days, based on the expected overflow of 60 ft^3/day. However, hydraulic detention times occasionally dropped to 10 days, and Windt _et al_. (85) comment that some of the apparent reduction in oxygen demand may be due to dilution. Complete odor control was achieved since the ditch always had ample dissolved oxygen present. Power requirements were reported to be 60 kwh/day. No indication was given of the ultimate destination of the ditch liquor but since this contained a COD of 16,000 mg/l (just before marketing) it would seem further treatment would be essential.

Most of the literature is occupied with reports on underfloor oxidation ditches. Though Pontin and Baxter (58) and Smith and Hazen (71) have reported on external ditches, there is little published information on the effect of low temperatures upon treatment efficiency for livestock wastes. Grube and Murphy (25) have described the operation of a ditch for municipal wastes in Alaska. The rotor was covered but the ditch was open. The minimum ditch temperature observed was 35.5°F (1.9°C) but BOD_5 removal during the winter period exceeded 90%. They practiced complete sludge return, but they observed that auto-induced sludge discharge took place intermittently. The hydraulic detention time was 2.3 days and the average MLSS was 2000 mg/l. These results are very different from those reported for agricultural wastes, though rather typical for municipal wastes.

Renovation by Soil Percolation

Although the treatment systems described by Stander and VanVuuren (72) and by Culp (18) are capable of providing potable water, such systems are not presently applicable to agriculture. Fortunately, using the soil itself for wastewater treatment can result in an effluent approaching drinking water quality, as has been demonstrated by man's successful use of wells for generations. Until recently, groundwater recharge after soil percolation has been unplanned, or subsidiary to other water use such as irrigation or the disposal of septic tank overflow.

McGauhey and Krone (50) have compiled a very comprehensive review of the literature associated with the soil mantle as a wastewater treatment system and their work will be referred to extensively in this report. McGauhey and Krone (50) developed their review from preliminary work with septic tank disposal fields because they found that engineering guidelines were sparse, or only applicable regionally.

The soil may be described qualitatively as comprising four distinct components, inorganic solids, organic solids, aqueous solutions and gases. Soil can exist in two states, aerobic and anaerobic. A fertile soil which sustains a thriving crop is aerobic since plant roots require oxygen. A completely waterlogged soil containing organic matter and suitable microflora will generally turn anaerobic. McGauhey and Krone (50) examined the action of the soil on wastewater percolating through it and they cite the literature as showing that soil maintained in an aerobic condition will be able to remove more organic matter from the liquid applied than anaerobic soils, moreover aerobic soils will retain their ability to transmit water longer than anaerobic soils. Two reasons for anaerobic clogging are cited: either that facultative bacteria seem to use the wastewater to generate slime capsules of polysaccharides which block the soil pores, or sulphur compounds are reduced to insoluble metallic sulphides which cement the soil particles together blocking further water transmission.

The movement of water through soil is governed by the rate at which water may enter the surface, called infiltration, and the rate at which water is conducted through the soil, called permeability. Unlike some engineering parameters neither infiltration rate nor permeability are immutable properties of the soil type. Continuous inundation of the soil surface will ultimately decrease the infiltration rate because of anaerobic bacterial growth clogging the soil pores. Allison (3) conducted tests using sterile soil and sterile irrigation water and proved conclusively that after steady state conditions (with regard to air entrapped in the soil) were reached, soil permeability remained constant. Although Allison (3) talks about permeability he also mentions that the maximum resistance to flow in nonsterile soil occurs in the top 12 in., hence it is the infiltration rate which is affected by the microbial growth.

The decrease in infiltration rate of inundated soil has led to two approaches for groundwater recharge. Bouwer (10) has reported on the Flushing Meadows project in Arizona using six parallel recharge basins with grass, gravil or soil surfaces. Decreasing infiltration rates were overcome by using a cycle of 3 days wet and 4 days dry; in this manner, infiltration rates around 24 ipd per ft of water depth could be maintained. The other approach, that of spray irrigation, has been used by Flower (21) for disposing of a high strength drinking waste. Sprinklers were set up on 30 ft towers above a 30 ac wooded hillside, the irrigating schedule was 8 hr on and 48 hr off. Some runoff was observed but this was small.

The object of any system of soil percolation is to ensure that the percolate is inoffensive when it reaches the groundwater. In the past this has merely required that the carbonaceous matter and the suspended solids needed removal. However, recent concern over the eutrophication of lakes has led to investigations into the rate of N & P during passage through the soil.

Carbonaceous material is removed very readily if the soil is allowed to become aerobic intermittently. Flower (21) reports that springs fed from percolate that had passed through 9 ft of sand under forest floor litter showed a BOD_5 = 10 mg/l, when the applied effluent was 400 mg/l. Bouwer (10) found, using secondary effluent from a municipal treatment plant, that a BOD_5 = 0.2 mg/l could be obtained after soil percolation. But the soil may not always be able to renovate the organic load adequately if it is held in an anaerobic condition; Stewart et al. (73) found that the groundwater under actively used corrals in Nebraska showed an average soluble carbon contents of 73 mg/l compared with 14 mg/l found under irrigated fields, moreover, these corral samples were noted as having an offensive odor suggesting anaerobic conditions.

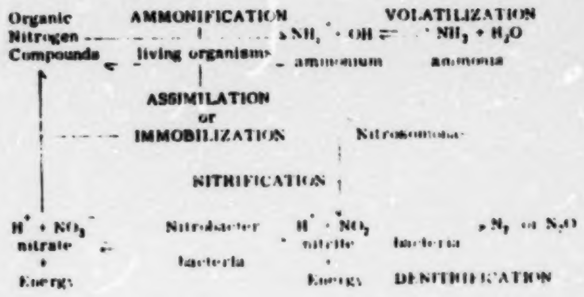

Organic
Nitrogen
Compounds
AMMONIFICATION
living organisms
$NH_4^+ + OH^-$
ammonium
VOLATILIZATION
$NH_3 + H_2O$
ammonia
ASSIMILATION
or
IMMOBILIZATION
Nitrosomonas
NITRIFICATION
$H^+ + NO_3^-$
nitrate
+
Energy
Nitrobacter
bacteria
$H^+ + NO_2^-$
nitrite
+
Energy
bacteria
N_2 or N_2O
DENITRIFICATION

Figure 1. Transformation of nitrogen in soil

The application of nitrogen to soil can lead to a variety of effects. Frederick and Broadbent (23) show that microorganisms cause a variety of reactions as shown in Figure 1. The agronomist has been most interested in retaining N in the soil for plant use but the engineer is more interested in removing N before it can accumulate in groundwater. Reference to Figure 1 shows that if a readily available carbon source is present some nitrogen is converted into cell matter in the microorganisms using the carbon as a substrate. However, as the carbon source is depleted, if aeration and temperature conditions are favorable, then the nitrogen will be converted to nitrate via autotrophic bacteria. Nitrate is not readily absorbed by soil particles or organic matter, hence the nitrate will readily move down into the soil with leaching water. Bremner and Shaw (11, 12) examined the reaction of nitrate in considerable detail and determined that if conditions became anaerobic, and an available carbon source was present, most nitrate would be converted to gaseous N_2O and N_2. Some very small conversion to NH_3 was determined, but this was considered trivial. Temperature, pH, and degree of saturation were all factors affecting rate of denitrification. Over a 20 day period little denitrification occured if the temperature dropped below 41°F (5°C). The percentage of denitrification obtained was the same for all soils in the pH range 5 - 8. If the soil water content exceeded 200% of field capacity then denitrification was rapid. Aeration of soils below 70% of field capacity inhibited denitrification completely.

Parizek et al. (55) have described extensive renovation studies using sprinkler irrigation. Using 1 or 2 in./wk application rates on woodlands, they found that 68 to 82% of the nitrate nitrogen in a secondary sewage effluent was removed by the first 12 in. of soil. Four in./wk was also tried but it gave inferior removals of N. These tests were conducted from June to December in Pennsylvania.

Meek et al. (44) have reported a field laboratory study using lysimeters fitted with drain tiles. The lysimeters were 120 in. in depth and 15½ in. dia and the tiles were fitted at 76 in. - 96 in. and 120 in. depths. The tiles could be held below the water table by means of a syphonic discharge tube. The lower part of the columns were maintained in an underground laboratory held at 77°F (25°C) but the top 3 in. of the column was exposed to the atmosphere. All columns were planted with Safflower (Arthamus tinctorius L.) and the water table in all columns was held at 70 in. depth. The irrigation water was spiked with NH_4NO_3 during the first and fifth irrigation to give 100 lb/ac. Curves are presented that show the nitrate concentration to peak at 28 in. but to drop severely at the tile outlets. Denitrification increased as the depth of the tile chosen for sampling. Soluble carbon was shown to be present at all depths but the level decreased with increasing total irrigation amount. Thirty seven in. of water were applied to each column in the period October to January at approximately two week intervals; only 40% of the water applied was recovered in the drainage. Although DO was shown to be present at all depths, it was usually less than 1 mg/l. The soil redox potential declined from 400 mv at the water table to 0 mv at 120 in. depth. Meek et al. (44) conclude by suggesting their results show that periodic drying increases soluble carbon in the soil solution and this should promote denitrification.

Another element of interest to engineers is phosphorus since this has been implicated in the eutrophication of lakes. An early 10-yr study by Morgan and Jacobson (46), using lysimeters planted with tobacco, showed no phosphorus leaching. Coleman (16) examined the P adsorption of kaolinite and montmorillonite clays and showed that, although pure clays with Fe and Mn removed would adsorb PO_4^{-3}, twice as much would be retained in the presence of Fe & Mn; however, such retention was then a function of pH. More PO_4^{-3} was retained at pH 3 than pH 9.5. Cooke (17) has examined P & K mobility in soils and finds that PO_4^{-3} ions are very immobile, to the extent that often more than 50% of applied P fertilizer will remain in the soil after harvest. He also concludes that in most cases soluble phosphate fertilizer are better than less soluble forms. Retention of P by clays is sometimes sufficient to cause large removals in the soils percolate

(Parizek et al. (55)) however Johnston et al. (33) have noted some anomalies since they found appreciable quantities (1 - 17%) of applied P in irrigation wattr, or fertilizer, could appear in the tile drainage effluent. The soils were Californian Panoche and Oxalis silty clays which are permeable basin rim soils.

Summary

Wide spread water reclamation and reuse is not only desirable, it is imperative. By changing from straight through to recycled flow, one establishment, or a complex using coordinated reuse, will use less fresh water and also discharge less wastewater.

Hydraulic handling and transport of livestock wastes has much to offer in terms of efficiency and labor savings. It does, however, employ large volumes of liquid. Thus agricultural producers can learn much from the manufacturing industries, although equally elaborate systems should not immediately be necessary. For example, the renovated effluent which is to be used for hydraulic manure transport need not be wholly free of organic matter. But some requirements which do emerge from the literature are: the liquid must not smell bad; it should be easy to handle using standard equipment; and biological, or other, treatment should be low in cost. The most frequently recommended treatment system that satisfies all three requirements seems to be the extended aeration process, in the configuration of the oxidation ditch. The literature reveals, however, that design data for treatment of livestock wastes by this method are currently inconsistent, and often seem arbitrary. Odor control has been of more concern than has final disposal of the treated waste. Consequently information concerning sludge production rates and the problems of handling this material is sparse. Nutrient imbalance is of concern to agronomists, but little information is available on the levels of nutrients which may be established in a recycling system.

Any excess water from the treatment system immediately associated with recycling may require further treatment before release to groundwater or a watercourse. Spray irrigation should be possible during the warmer period of the year when the soil is thawed. The literature reviewed has shown that C, N & P are all decreased during soil perculation and the effluent is very similar in quality to existing groundwater. Spray irrigation would seem more promising than ponding or ridge and furrow application since accurate grading of the terrain is of less importance and control of application rate more easily achieved.

OBJECTIVES

Of the various methods for keeping low odor levels in a confinement livestock buildings, two stand out. Either the manure can be aerobically treated within the building to eliminate septic odors, or the manure can be continually removed from the building before it can putrefy. The main objection to the first alternative is that restarting the aerator, following a prolonged stoppage, may release sufficient toxic gases, such as H_2S (Taiganides and White (55)), to kill the animals in the house. Hence, Iowa State University has elected to investigate external treatment systems. Early disappointments with a mechanical manure removal system led to the adoption of hydraulic transport. The work described in this report is one segment of an ongoing project in which the basic parts of the system already established were:

A 700 head environmentally controlled swine finishing building
A functioning anaerobic lagoon
An existing, but unconnected oxidation ditch

A preliminary study by Smith (70) had shown the feasibility of treating the supernatant from the anaerobic lagoon and reusing this treated effluent for hydraulic manure transport. The objective of the present study was to develop a better, and more reliable, system for recycling the treated effluent. The parameters selected for monitoring were mainly chemical and were related to water quality. Odor is generally associated with anaerobic or septic conditions, thus oxygen demand was expected to indicate the potential that the treated effluent still possessed for putrefaction. Both COD and BOD_5 were to be measured because their ratio gives some indication of the stability of the organic matter in the liquid. Solids content of the liquids would give useful information on volatile solids reduction, sludge production, solids retention time and the rate of buildup of dissolved inorganic matter. Measurements of nitrogen forms and phosphate were made to find the fertilizer potential of the effluent. The chloride ion is generally accepted as a useful inert tracer in flow studies; hence this was chosen for detecting changes in concentration of inorganic salts as recycling progressed. Total coliform counts were made to indicate the sanitary quality of the returned effluent. Other measurements of interest were temperature, water flow and power use.

The excess water contributed by waterer spill, feces and urine was stored in an anaerobic lagoon. This lagoon water was still a potent source of pollution and could not be released to a water course without further treatment. Rath (61) had tried this anaerobic lagoon effluent in laboratory lysimeters using the local prevailing soil type (Webster silty clay loam) and found reductions in COD could be obtained in as little as 2 in. of soil, if the lagoon liquor was applied for an interval exceeding 8 hr in 24 hr. However, Rath (61) only ran each lysimeter for 4 days so his results are inconclusive with regard to quality renovation.

Thus, the irrigation portion of this research was designed to determine the degree of removal of C, N and P as the liquid moved down through the soil. It was also necessary to relate applied quantities with soil moisture conditions. Finally, soil samples were to be taken at intervals to monitor the effect that treatment has on soil carbon and nitrogen. The study was to be conducted over several irrigation seasons to determine if use of the soil in this manner has any long-term detrimental effect.

SYSTEM AND ITS OPERATION

Hydraulic Circuit

The research was planned in two phases. In Phase 1 the manure slurry was flushed from the confinement house, hereafter referred to as Unit K, to an anaerobic lagoon (Figure 2). The ditch was supplied from the lagoon and settled effluent from the ditch was then returned to flush tanks in Unit K. Minor trouble with plugging occurred in the 8 inch sewers connecting Unit K and the lagoon, these had too many abrupt changes in direction and insufficient slope.

Because anaerobic lagoons are reputed to be evil smelling, Phase 2 diverted the manure slurry directly to the oxidation ditch. Gravity flow was not feasible so Pumps 6 and 7 were installed. A lagoon was still essential to receive the overflow from the oxidation ditch.

Unit K

Unit K has an average capacity of 700 finishing pigs. It is a totally enclosed steel frame building, 50 ft wide by 120 ft long. The floor slopes at 1 ft in 120 ft in the longer dimension. The pens are nominally 5 ft wide and are spaced each side of a central alleyway. The pen floors slope at 1 ft in 24 ft from the center alley to a gutter 42 in. wide by 1½ in. deep. The two gutters are adjacent to the longer walls of the house. The pigs are floor fed and have free access to the section of the gutter in their pen. The liquid used for flushing is held in a tank, supported 8 ft above the floor, at the upper end of each gutter. Each tank can discharge

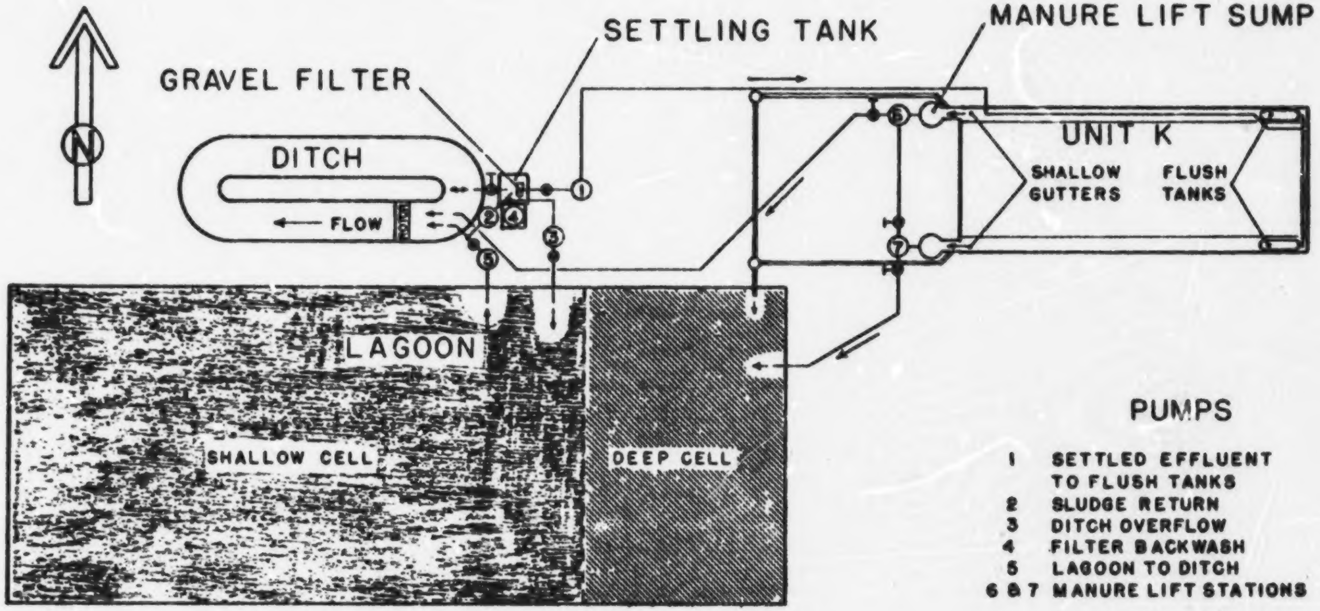

GRAVEL FILTER
SETTLING TANK
MANURE LIFT SUMP
N
DITCH
FLOW
UNIT K
SHALLOW GUTTERS
FLUSH TANKS
LAGOON
SHALLOW CELL
DEEP CELL
1
2
3
4
5
6
7
PUMPS
1 SETTLED EFFLUENT TO FLUSH TANKS
2 SLUDGE RETURN
3 DITCH OVERFLOW
4 FILTER BACKWASH
5 LAGOON TO DITCH
6 & 7 MANURE LIFT STATIONS

Figure 2. Hydraulic circuit, Unit K

up to 120 gal in 20 sec, this discharge is initiated by a time switch.

Anaerobic Lagoon

The lagoon has two cells; the first cell is about 8.8 ft deep and has a volume of 40,700 ft^3, the second cell being a shallow extension of the deep cell with a depth of about 3.05 ft and a volume of 46,300 ft^3. Only approximate depths can be given since the lagoon serves as storage and thus varies in depth. This lagoon and its operating parameters have been described by Willrich (84).

Oxidation Ditch

The oxidation ditch is a simple oval with a single Passveer rotor. A plywood cover was installed over the whole ditch in 1969 because intermittent use of the ditch in winter had led to icing difficulties. The cross section is trapezoidal and is lined with concrete slabs. The working volume of the ditch, as measured at the end of Phase 2 using a chloride dilution method, is 47,000 gal. Since this report is a continuation of previous research, the reader is referred to Smith (70) for more detail. The aeration characteristics of the ditch before lining are described in Knight (37).

Flush Tanks

Preliminary work had shown that a simple flap valve connected directly to a small solenoid would not function reliably. The initial rush of water down the discharge pipe just slammed the valve straight back on its seat; then this opening and closing cycle would continue rythmically until the tank was empty. A mechanism was developed (Figure 3) in which a large expanded polystyrene foam float provided the lifting force for the valve and the solenoid only operated a latch mechanism. This device has worked well once suitable construction materials were chosen. The float rod is brass and slides in a nylon bush. Most of the latch pivots are small sealed ball bearings. The two tanks are identical and are adjusted to contain the same quantity of liquid.

Lagoon to Ditch Pump. Pump 5

Previous experience (Smith (70)) with a helical rotor pump mounted at ground level was not satisfactory. This system was replaced by a wet well consisting of two 4 ft vertical sections of 5 ft dia concrete pipe. The wet well is connected to the lagoon by a length of 4 in. fibre pipe, valved to facilitate maintenance. A domestic sump pump, with a ¼ in. mesh wire fabric screen, was used to pump lagoon supernatant from the wet well into the ditch. This pump was controlled by a float switch mounted in the settling tank.

This pump functioned until May 1969 and then burned out. When inspected the rotor and pump body were found covered in a dense layer of gray crystals (Figure 4). These were later analyzed as $Mg(NH_4)PO_4$. Rawn et al. (62) had reported this material would plug pipes in anaerobic digesters. They mention that removal of CO_2, causing an increase in pH, would cause deposition. The material has since been found wherever anaerobic lagoon liquor is passed through a pump or small constriction. $Mg(NH_4)PO_4$ does dissolve in acid but it does not dissolve appreciably in hot water.

Glossy photos attached

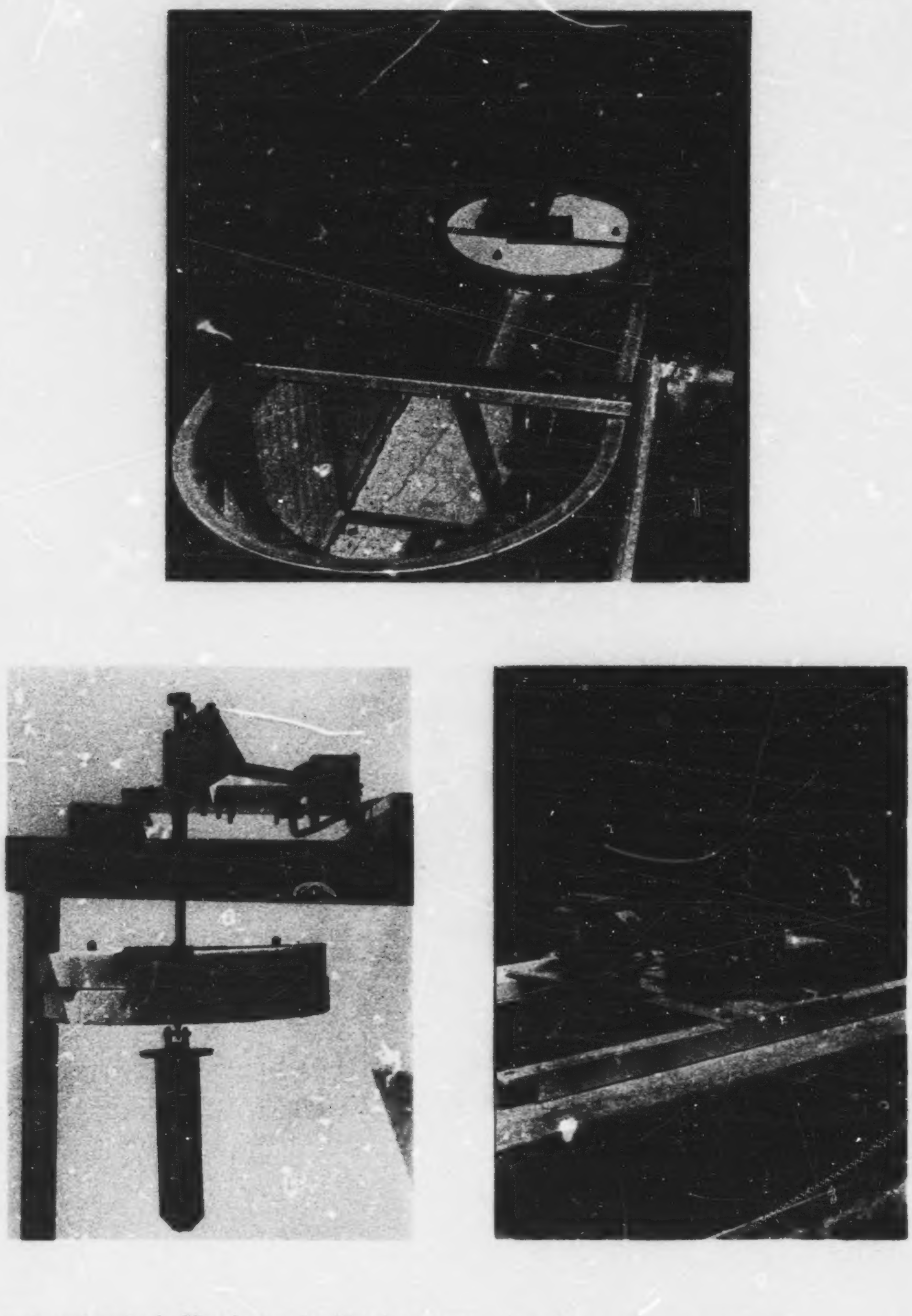


Figure 3. Float actuated flush tank discharge mechanism. Top center. Total assembly with the dust covers removed. Bottom left. Float and flap valve latched in upper position by solenoid. Bottom right. Valve mechanism in lower, sealed position.

Glossy figures attached,

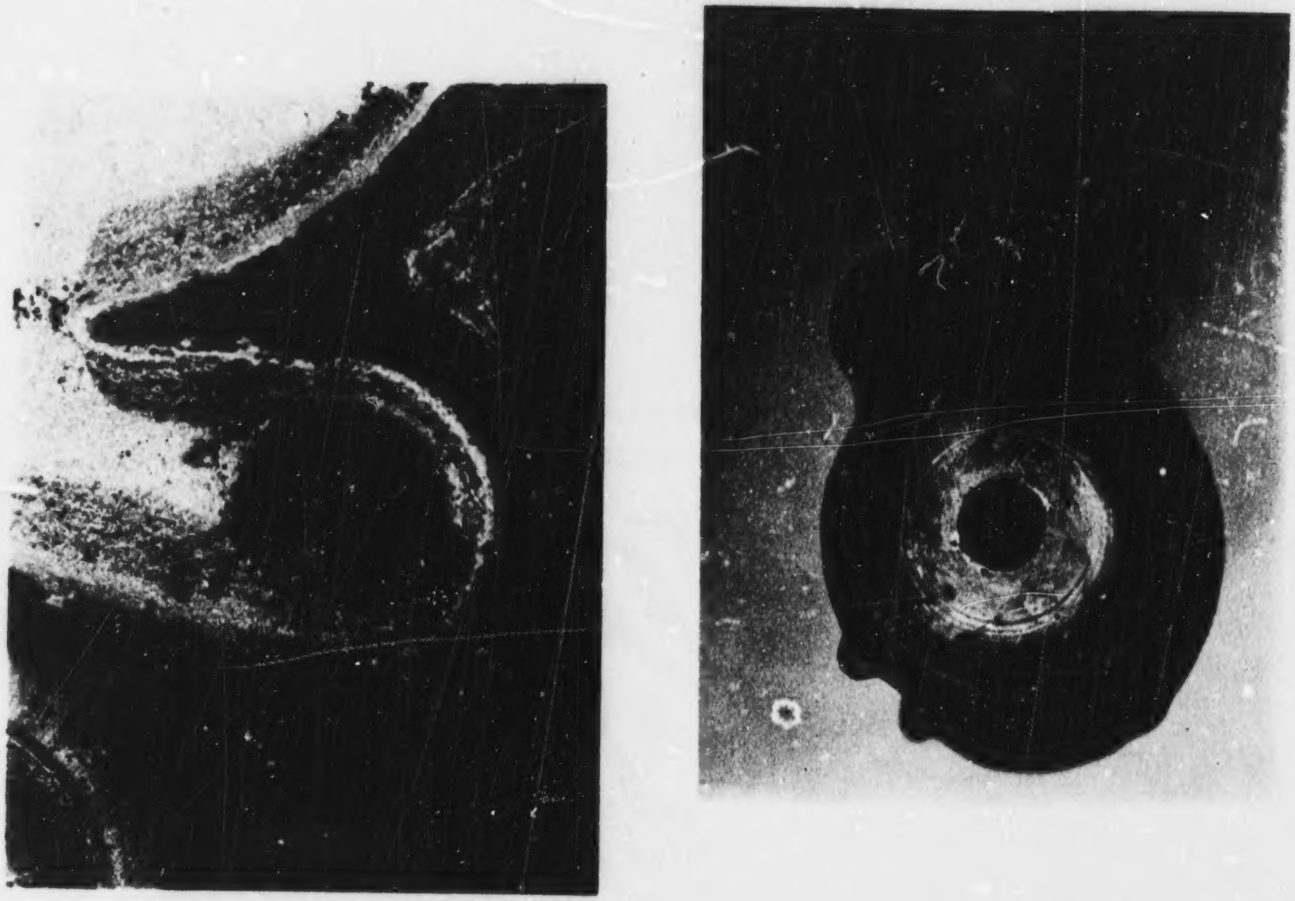


Figure 4. $Mg(NH_4)PO_4$ deposits on a pump after being submerged in the lagoon for six months.

Rotor Drive Unit

The original rotor drive unit used in prior studies (Knight (37) and Smith (70)) was a variable speed drive coupled to the rotor by a single strand 1 in. pitch chain. Phase 1 continued to use this unit since the depth of immersion was not great (7.5 in.) and the speed was only 75 rpm. However, after Phase 1 was finished, it was found that the output drive shaft bearing of the variable speed drive unit was badly worn. Since greater oxygen transfer was required in Phase 2, the final speed of the rotor was to be increased to 120 rpm. Because this was the second failure of the bearing, it was decided to replace the variable speed unit by a simpler, more robust, unit. This had a jack shaft running in roller bearings, belt driven from a 900 rpm motor. Some changes were also made in the chain case in an attempt to control lubricant loss.

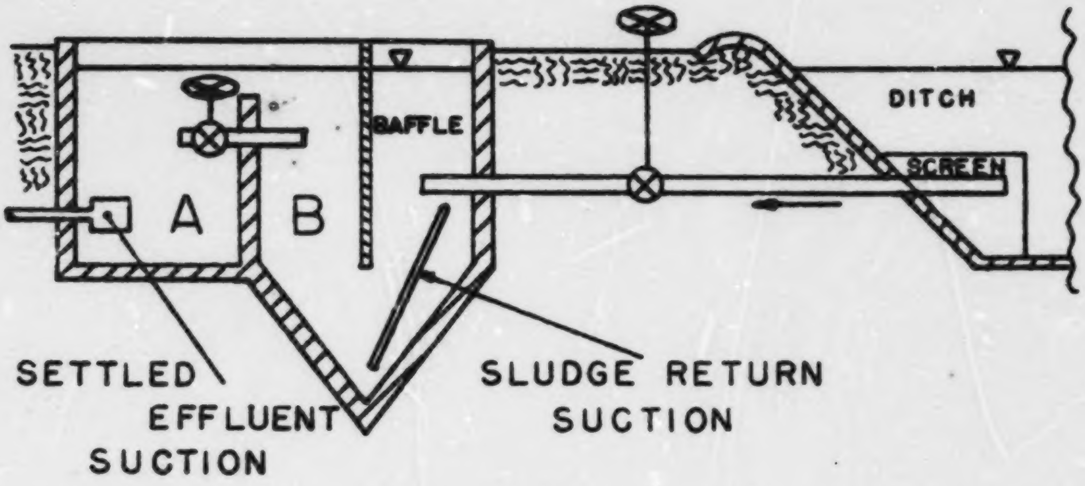
As operation was about to begin in August 1970, it was discovered that the rotor bearings were also badly worn. These were replaced at short notice with bearings fitted with incorrect grease seals. This deficiency was overcome during the test period by liberal daily greasing. Chain lubricant was lost from the final drive case, partially as a consequence of the incorrect seals, hence wear on the chain drive has been rather severe. The author has observed the rotor manufacturer's design changes over the years, and it seems that drive train longevity is a problem in cage-type rotors. Robinson *et al*. (65) have reported a completely new design of rotor consisting of perforated discs, which they say show none of the drive train problems of the Passveer design. Moreover, they claim less horsepower is required for the same oxygen transfer rate, particularly at high immersion depths and high speeds.

Settling Tank and Sludge Return

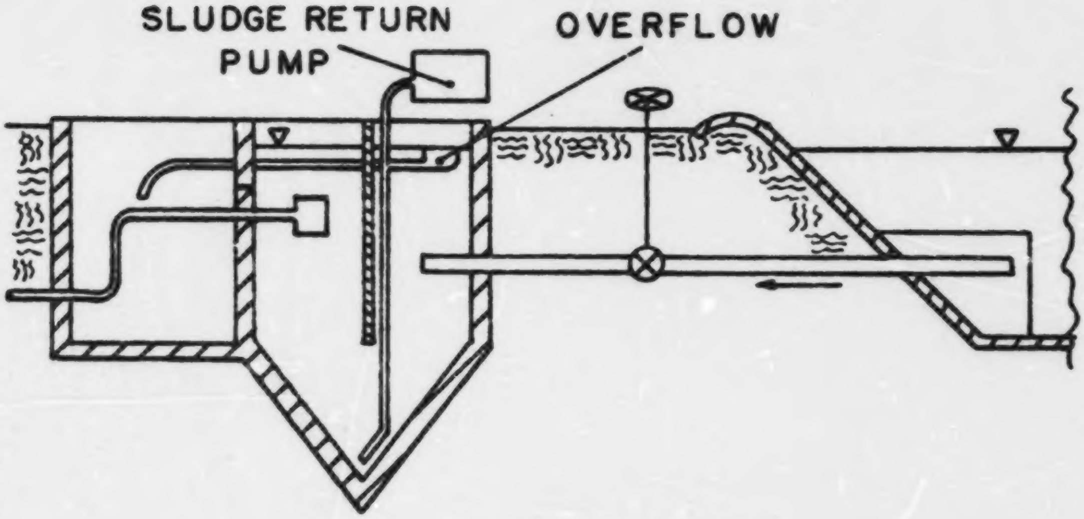
The settling tank shown in Figure 5 has a plan area of 5 ft x 5 ft, is 8 ft from the top to the apex of the hopper bottom, and is divided into two parts by a baffle. In the Phase 1 plumbing prior to May 1969, the liquid flowed through compartment B and into compartment A. Compartment A was to function as a clear well for the suction of the effluent return pump (Pump 1). The sludge was pumped back into the ditch by Pump 2. Two major problems were encountered with this configuration: first, compartment A began to fill with the fine material not settled out in B, ultimately choking Pump 1; secondly, the rather tortuous suction to the sludge return pump plugged causing this pump to run dry.

These problems were largely overcome in Phase 2. The effluent return pump suction was brought up into compartment B and A was blanked off by extending the dividing wall. The sludge return pump was remounted over compartment B with a more direct suction line. A further feature which had to be incorporated was an overflow weir. This was fabricated from an inverted 2 in. condulet box. The overflow drained into compartment A and was then pumped into the lagoon by a domestic submersible sump pump (Pump 3).

In spite of these modifications the system did not function according to plan. The much higher solids level, established in the ditch with the raw manure influent, produced a liquor that was more in the nature of a fluidized carpet than a liquid. The hair, husk and dense biological floc


BAFFLE
DITCH
SCREEN
A
B
SETTLED EFFLUENT SUCTION
SLUDGE RETURN SUCTION

TO MAY 1969
(PHASE 1)


SLUDGE RETURN PUMP
OVERFLOW

AFTER JULY 1969
(PHASE 2)

Figure 5. Settling tank for oxidation ditch effluent

conspired to plug both the 1¼ in. suction to the sludge pump and the 1¼ in. overflow line. Moreover, when the suspended solids level in the ditch rose above 12,000 mg/l, no appreciable solid/liquid separation took place, and the hairy liquor plugged the suction strainer to the effluent return pump.

An open topped box, partially filled with a 3 in. layer of 3/8 in. gravel, was attached to the suction of the effluent return pump. Initially this filter could be manually backwashed with fresh water, but later it was connected to an extra pump (Pump 4) which could automatically backwash it with settled effluent. By late January solid/liquid separation was so poor that all screening was abandoned and a different pump was substituted for effluent return.

After running dry for the second time, the small helical rotor pump used for sludge return was abandoned. The initial replacement was a domestic sump pump but it, too, plugged with hair, and polyethylene film thought to be from rat bait bags used in Unit K. Finally a diaphragm pump, fitted with 3 in. suction line, was installed and no further problems were encountered.

Effluent Return Pump. Pump 1

Phase 1 used a conventional clear water centrifugal pump controlled by a pressure switch responding to closure of the float valves in the flush tanks. This pump worked with little attention, except for periodic air replenishment in the surge chamber.

For Phase 2 a new surge chamber was installed. This had a diaphragm separating the liquid and air compartments and proved more satisfactory. The centrifugal pump was regulated to deliver against 32 psi head. At this pressure, the pump was rated at 40 gpm. This produced an average upflow rate in the settling tank of 460 gpd/ft^2. However the tank was small, and the positioning of baffle such that local upflow rates may have been much in excess of 460 gpd/ft^2. The poor liquid/solid separation was attributed to excessive upflow rates. The Water Pollution Control Federation (81) does not recommend a rate in excess of 800 gpd/ft^2 for small final settling tanks. This view was further reinforced when it was found that the sludge interface was at least 24 in. below the water surface after two hours of quiescent settling. The centrifugal pump was replaced by a low speed flexible impellor pump which was rated at 3.5 gpm. Although the results were at first encouraging, the sludge blanket would eventually rise and plug the gravel filter. One favorable feature of the flexible impellor pump is that it will pump some solids, so, for the last two months of the test, no screening was practiced and the material was simply returned to Unit K with the consistency of thin soup.

To avoid problems with the thicker fluid, the plumbing of the flush tanks was altered so that they were bottom fed through their drain plugs. The float valves thus were bypassed and filling was controlled by a time clock. The liquid levels in the two tanks could equalize through a connecting pipe, maintaining approximately equal discharge from each tank.

Manure Lift Stations

As mentioned earlier, Phase 2 required that the manure be pumped from Unit K to the ditch. The design of pumping facilities was affected by the existence of two 8 ft dia and 8 ft deep manure storage pits, hence it was decided to build two separate, but identical, pumping stations. A wet and

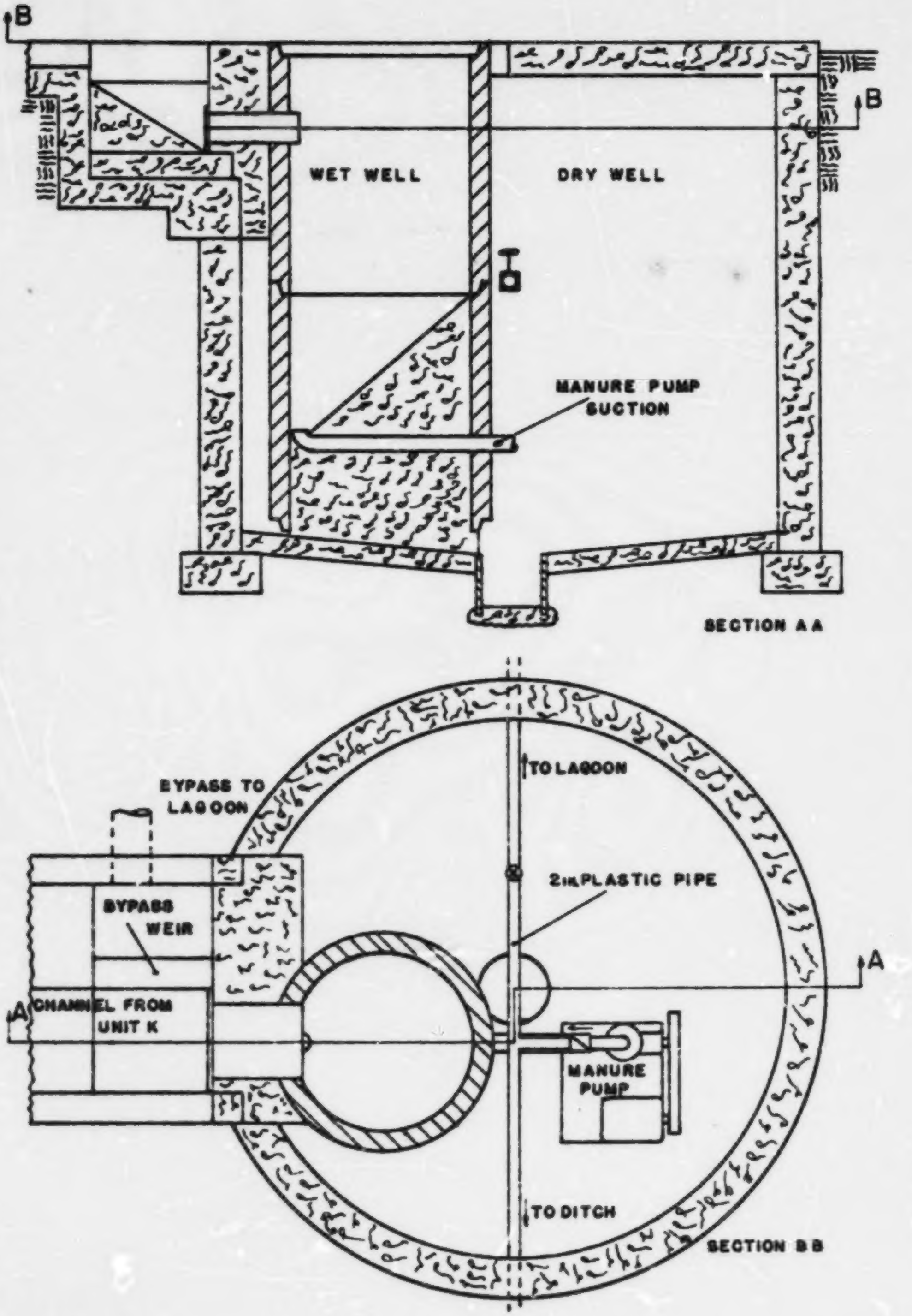

B
B
WET WELL
DRY WELL
MANURE PUMP SUCTION
SECTION A A
BYPASS TO LAGOON
TO LAGOON
BYPASS WEIR
2in PLASTIC PIPE
A
A
CHANNEL FROM UNIT K
MANURE PUMP
TO DITCH
SECTION B B

Figure 6. Manure lift stations

dry well construction was chosen as shown in Figure 6. Helical rotor pumps were selected because they handle solids quite well and are capable of developing the high discharge heads needed should plugging occur in the discharge line. Submerged suctions were used and discharge was to a 2 in. dia polyethylene force main. Valving was provided so that both stations could discharge either to the lagoon or to the oxidation ditch. This is shown more clearly in Figure 2. Raw manure slurry contains a substantial fraction of readily settled material, hence the wet well was constructed with a sloping floor and the pump suction was located at the lowest point. Upon flushing, manure entered through a duct high on the side of the wet well and this gush of material proved very effective in scouring any residue from the prior pumping.

Since flushing is intermittent, this was capitalized upon in the design of the level control as shown in Figure 7. The switch has neutral bouyancy; it hangs vertically down in air but rolls to a horizontal position when immersed. The two way mercury switch has two functions. When the liquid level rises the float rolls over and the pump turns on. Then as the level falls, the other contact starts a delay timer which is set so that the pump stops after draining the wet well completely. The delay timer resets to zero automatically on completion of the cycle. This rather elaborate system was designed to ensure that the minimum quantity of solids are left on the bottom of the wet well between flushes.

A simple 2 x 1 in. wire mesh screen was placed in the duct to catch hard debris such as pieces of concrete, bolts and nails before they could injure the pump. This screen plugged within two days with a dense mat of

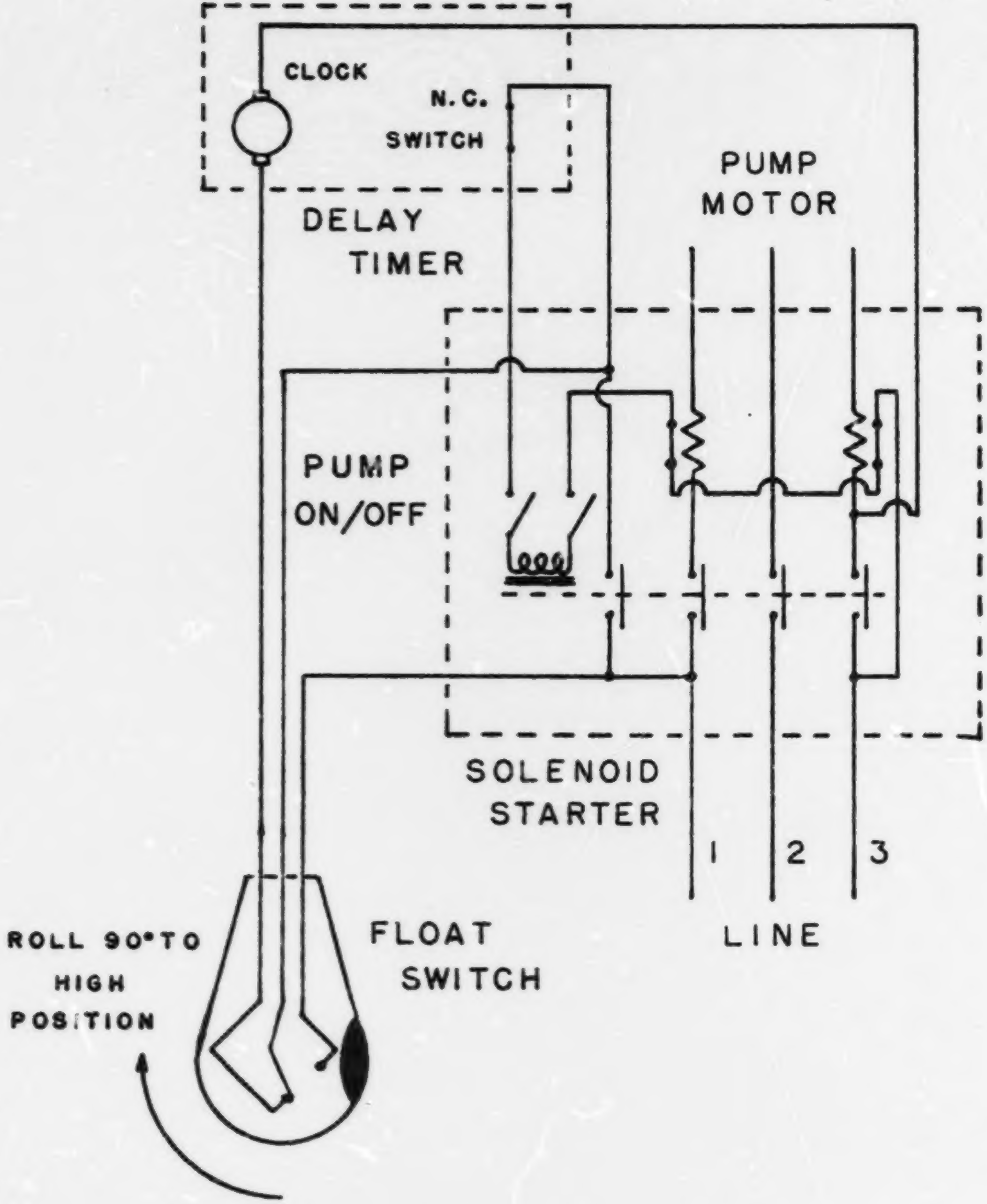

CLOCK
N.C.
SWITCH
DELAY
TIMER
PUMP
MOTOR
PUMP
ON/OFF
SOLENOID
STARTER
1
2
3
LINE
ROLL 90° TO
HIGH
POSITION
FLOAT
SWITCH

Figure 7. Electrical schematic of the level control used to activate manure pumps.

hair. A ledge 2 in. high was placed in the duct and worked quite well as a trap for heavy debris.

Metering

During Phase 1 the quantity of liquid being pumped into the tanks was measured by two domestic water meters. No simple and low cost meter could be devised to measure the effluent slurry flow, although it was known that this was in excess of the tank discharge by the amount contributed by feces and waterer spill. A three phase wattmeter measured the power consumed by the rotor but the other motors were not metered.

At the start of Phase 2 the returned effluent pump, Pump 1, delivered through water meters, but these rapidly plugged with hair, forcing the flow to the flush tanks to be measured by a bucket and stopwatch. The delivery of Pump 3 was correlated with running time. Neither of these measures are likely to be very accurate. Motor metering was improved by switching arrangements which allowed the choice of metering the rotor alone or the rotor and Pumps 1, 2 and 3. Pumps 6 and 7 were not metered.

Irrigation Field

An area 220 ft wide and 550 ft long, located directly south of the lagoon, was used for the irrigation field. This area has been in continuous corn production since 1961; previous to that time it had been in rotation pastures for swine. The field of which this area was a part had been fertilized in the fall of 1967 with 120 lb/ac of N in anticipation of corn production again the spring of 1968. The stalks were chopped and plowed under in November 1968.

The soil of this area is Wisconsin glacial drift. In original form it is characterized by a generally level, to flat or even depressed topography with few natural drainage channels but frequent lakes, ponds and sloughs. Artificial drainage has changed the characteristics greatly. The soils in the irrigation area are of the Clarion/Webster complex. The subsoils are heavy in texture, high in clay and relatively impermeable. The lime content is usually high. A shallow water table is also characteristic of the area.

The Iowa Sprinkler Irrigation Guide (29) shows that Clarion-Webster soils may be expected to have an infiltration rate, when a cover crop is present, of 0.5 - 1 iph. Figure 8 shows the general layout of the irrigation area. The topography is well suited to sprinkler irrigation with a maximum slope of 3% in the northeast corner to nearly level in the southwest corner. A small dike was built on the north, east and south sides as shown to eliminate foreign drainage. No attempt was made to control surface drainage across the plots within the irrigation field itself.

The Plots and Tile Drains

The layout of the system used consisted of twelve plots 40 ft by 60 ft in a rectangular pattern (Figure 9). Each plot was provided with a 5 in. clay drain tile at its center, laid approximately 4 ft deep. The buffer area between the plots was 40 ft which gave a tile spacing of 80 ft. The two drainage dimensions were taken from the Iowa Drainage Guide (29) for the soils in question. The drains were deemed necessary to ensure drainage of the profile and to follow renovation of the effluent as it percolated through the soil. The tile lines drained to a sump where outflow measurements and sampling for quality could be done. The buffer area provided enough separation between plots so that percolating water on one plot would have no influence on tile flow rates on those adjacent unirrigated plots. The section of drainage line from each plot to its associated sump was not sealed.

Each of the three sumps received drainage from four adjacent plots. These sumps were 48 in. dia corrugated metal culverts with five connections. Four connections led to each one of the plots and the fifth led to the main drainage line. This main line drained into an adjacent surface drainage ditch. The sumps had poured concrete bottoms.

The tile drain system was installed in May 1968. The water table was penetrated in all plots, usually at 3 ft below the ground surface. The area had been tile drained previously and old lines about 2 ft deep were uncovered, these were sealed with a plug of clay. Because the new lines were now below the water table, considerable drainage was observed once the construction work was completed.

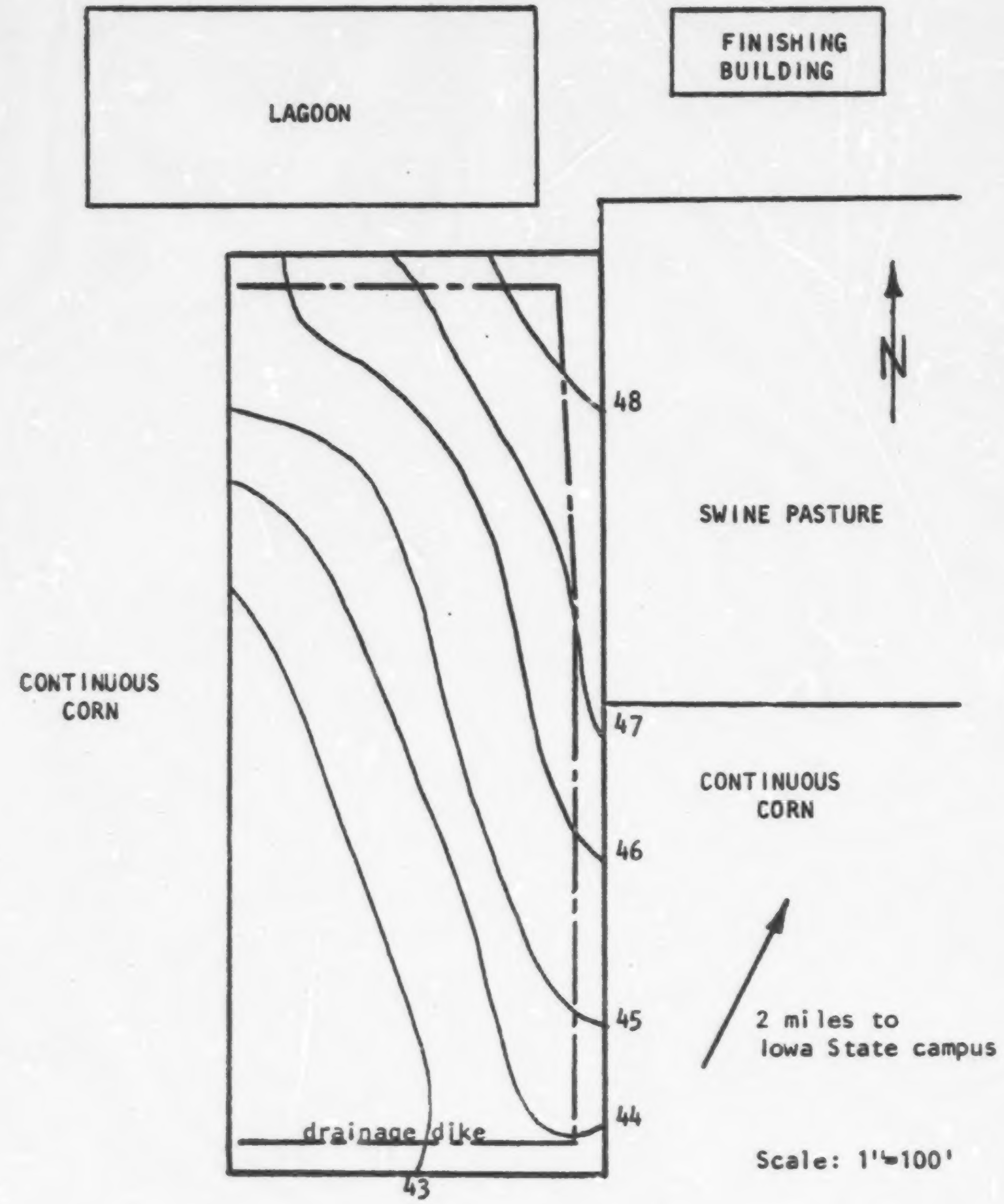
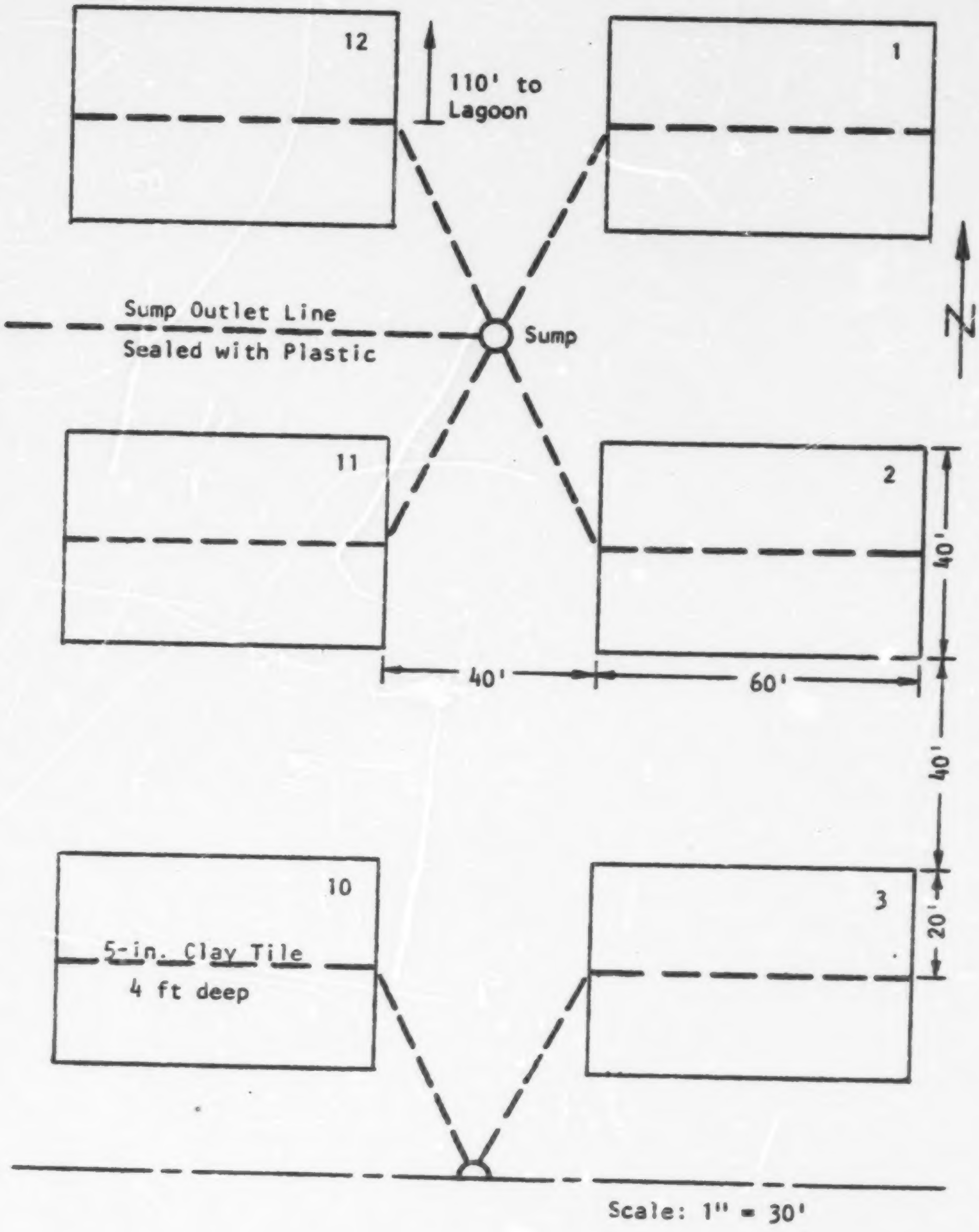

FINISHING BUILDING
LAGOON
48
SWINE PASTURE
CONTINUOUS CORN
47
CONTINUOUS CORN
46
45
2 miles to Iowa State campus
44
drainage dike
Scale: 1"=100'
43
N

Figure 8. General layout of irrigation area



12
1
110' to Lagoon
Sump Outlet Line
Sealed with Plastic
Sump
N
11
2
40'
40'
60'
40'
10
3
20'
5-in. Clay Tile
4 ft deep
Scale: 1" = 30'

Figure 9. Plan of one-half of plot and tile drainage system.

Intermediate Sampling Points

During the first season of irrigation, lysimeter pans, as shown in Figure 10, were placed at 3, 6, 12 and 30 in. depth at the center of each plot. The pans were embedded in undisturbed soil in the walls of the tile trench, the pans were partially filled with a layer of river sand before being placed in position. The pans drained into sample bottles contained in a 2 ft x 2 ft plan by 4 ft deep lined and covered pit. These lysimeters were similar to those used by Parizek et al. (55) in their study on the spray irrigation of secondary municipal effluent.

These pans were not entirely satisfactory since they required the soil moisture content to exceed field capacity before sufficient sample collected. In 1969 the plots were fitted with porous cup sampling points at 6, 12, and 24 in. As a check on the water passing below the tiles, two well points at 6 and 8 ft were also installed. Figure 11 shows the position of all sampling points in a plot.

Soil Preparation and Cover Crops

When drain installation was finished in 1969, the entire field was spring-tooth cultivated three times and then seeded to tall fescue with a Brillion seeder. However, because of the late seeding date of 10th May, the fescue had difficulty competing with indigenous giant foxtail. The foxtail was not a problem since the only interest was to establish a good cover crop.

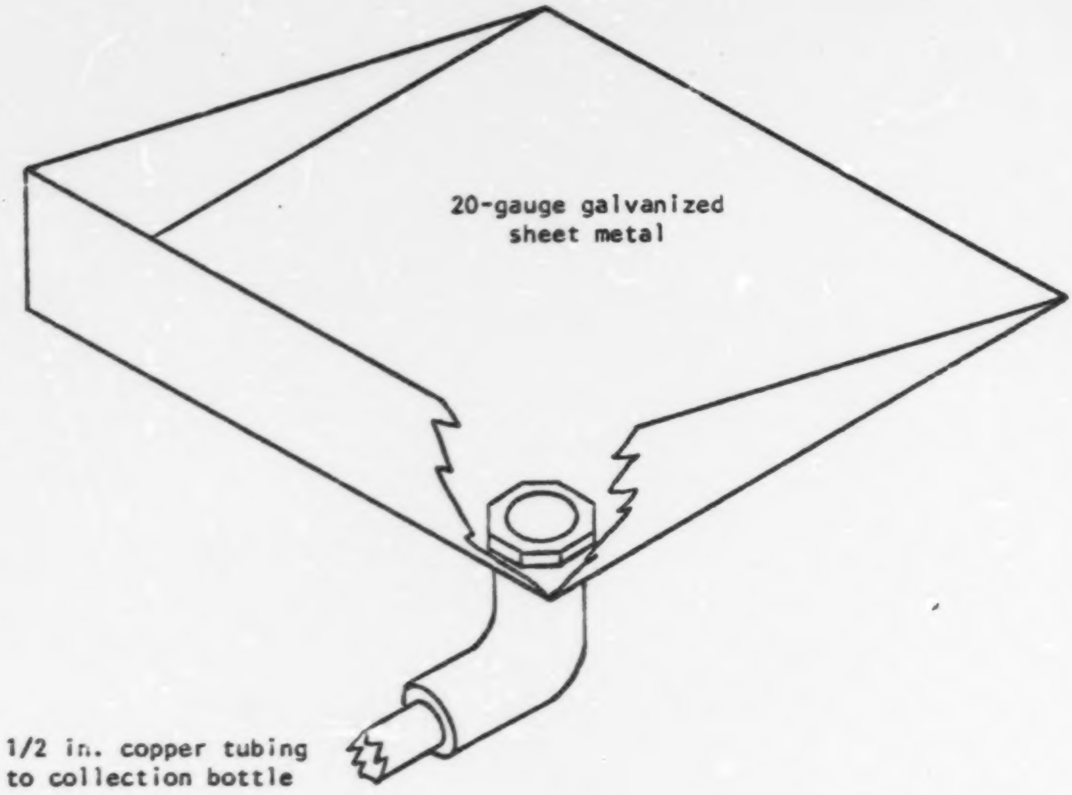

20-gauge galvanized sheet metal
1/2 in. copper tubing to collection bottle

Figure 10. Lysimeter pan

Table 1. Lysimeter pan sizes

Depth from Surface-in.	Pan Size-in.
3	6 by 6
6	6 by 6
12	9 by 9
30	12 by 12

Irrigation Equipment

An engine driven centrifugal pump was used for irrigation, this developed 20 - 45 psi at the required irrigation rates. An intake screen with 1/8 in. mesh was used on the pump suction to prevent large pieces of grass and other debris from entering the distribution system. The aluminium irrigation pipe was 3 in. dia for both main and laterals, all joints were fitted with quick couplers. A pressure gauge 20 ft downstream of the pump discharge was used to set irrigation rate.

Initially the system included six Rain Bird 25 TNT part circle sprinklers with 9/64 in. dia nozzles. These were located one at each corner of the plot, and are on each of the laterals halfway between the 0.44 - 0.70 iph, delivery at the ground depended upon line pressure and climatic conditions. Application efficiency would include about a 20% loss due to sideways spraying caused by the sprinkler turning arms plus loss due to evaporation. Later tests were run using only four sprinklers and only one line per plot. The location of these and the resulting water distribution is shown in Figure 11.

Irrigation Schedule

This section will discuss soil moisture criterion used for irrigation scheduling, the terminology use as by Baver (8). In 1968, soil moisture condition was determined gravimetrically. Shaw *et al*. (68) have examined Iowa soils with regard to their wilting point and their field capacity. Wilting point was defined as that moisture content at which the soil moisture tension was 15 atm and field capacity was determined for well drained field soils *in situ*. Using this data, 90% available moisture

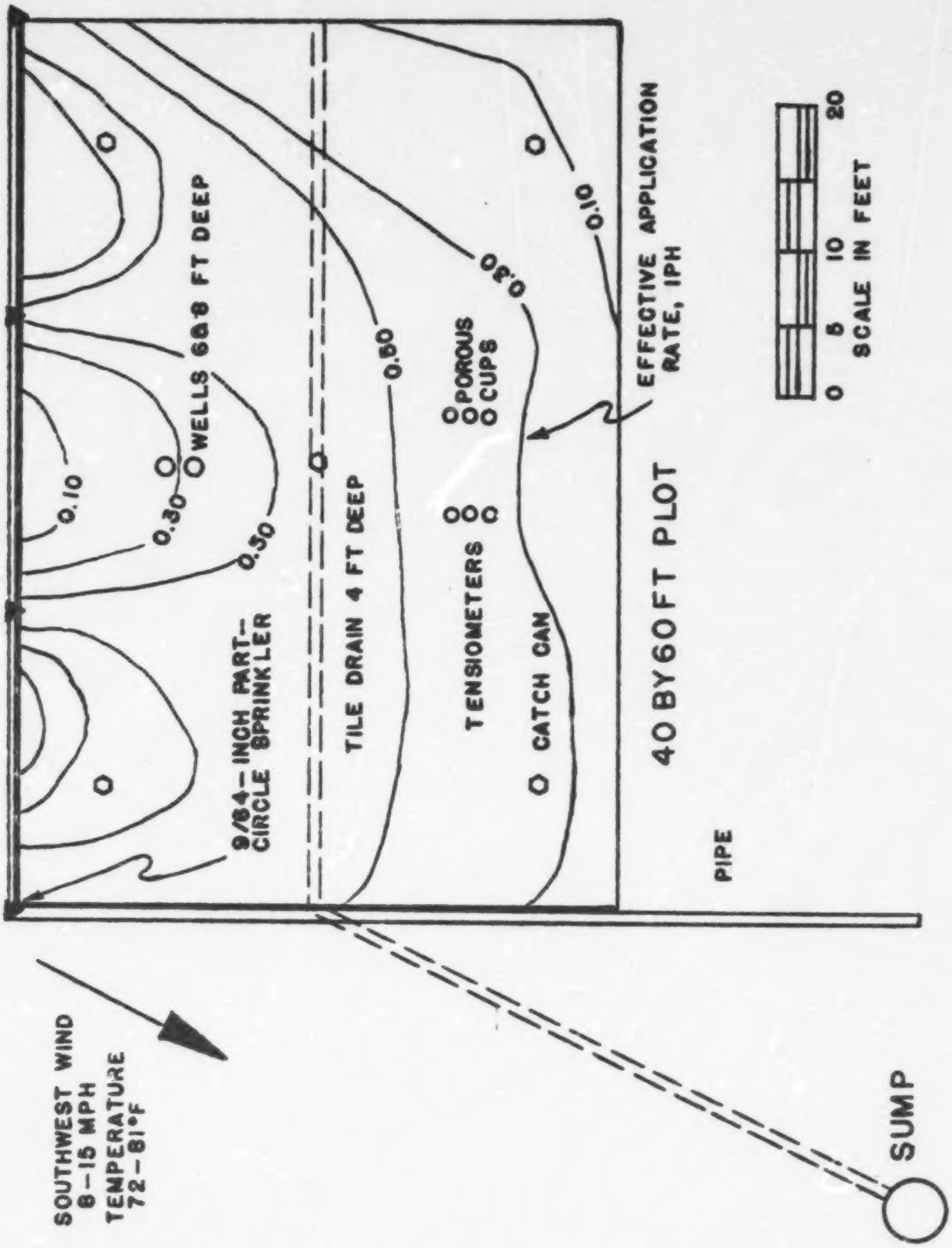

9/64– INCH PART–
CIRCLE SPRINKLER
WELLS 6&8 FT DEEP
0.10
0.30
0.50
0.50
0.30
0.10
TILE DRAIN 4 FT DEEP
TENSIOMETERS
POROUS
CUPS
CATCH CAN
EFFECTIVE APPLICATION
RATE, IPH
40 BY 60 FT PLOT
PIPE
0
5
10
20
SCALE IN FEET
SOUTHWEST WIND
8–15 MPH
TEMPERATURE
72–81°F
SUMP

Fig. 11. Experimental plot layout

(hereafter abbreviated as AM) corresponded to a soil moisture content of 29.0% (by weight) and 70% AM corresponded to 24.5%. Since gravimetric analysis required 24 hr for completion, soil tensiometers were installed in 1969. The criteria used with these instruments were; higher moisture content for irrigation corresponded to 350 mb (note 1000 mb = 0.987 atm) and lower to 750 mb. All the above moisture contents refer to the top 12 in. of soil.

In 1968 the four treatments used were; two levels of effluent amount, 1.5 and 3.0 in. per application, and two levels of soil moisture, 90% & 70% AM. Seventy percent represented a high criterion for conventional irrigation, while 95% might be more typical of a disposal field criterion. The lower application rate of 1.5 in. was a realistic minimum for irrigation set, while the upper application was near the maximum increment this soil profile could hold between 95% AM and saturation.

If the rate of application is compatible with the infiltration and percolation rates for a soil, application of lagoon effluent virtually assures that anaerobic conditions will be established in the profile for a time. This condition is caused by the increased biological activity engendered by the organic loading in the nearly saturated soil. The literature reviewed earlier (50) indicated that an aerobic rest period is almost mandatory so that infiltration rate may be recovered. Only irrigating after the soil achieves an available moisture level of 95% AM indicated that the soil has become aerated to at least the 12 in. depth.

For the next season it was decided to apply nitrogen in a known form and also to irrigate with clear water in addition to lagoon water. The four treatments were now equal in amount (2.0 in. per application) but different in quality.

By 1970 it had become apparent that single applications of 2 in. caused high transient concentrations of COD & N to reach the tile lines. Hence, the four treatments were now the same total amount but one treatment was 3 daily applications of 2/3 in., while the second treatment was one continuous application of 2 in. Two moisture tensions were used for each method of application. The three seasons' applications are summarized in Table 2.

Sample Collection

In 1968 samples from the tile under each plot, the lysimeter collection bottles and the surface catch pan were collected 12 hr after the beginning of each irrigation. Collection of samples caused by natural precipitation were collected 12 to 24 hr after the storm. The outflow rate from each plot was measured daily by the bucket and stopwatch technique. In addition, each sump was equipped with a V-notch weir and stage recorder that could record the continuous outflow from any one plot.

In 1969 and 1970 tile drainage water was sampled during each application of 2.0 in. and during the first and third application of the 3 x 2/3 in. daily applications. After the total 2.0 in. irrigation was complete, 4 more samples were taken at 2 hr intervals. The flow rate was recorded by bucket and stopwatch at each sampling so that a weighted composite sample for analysis could be made for each plot. Increased tile flow caused by precipitation was also sampled. This sampling procedure biased

Table 2. Overall summary of field experiments used to study soil filtration treatment of anaerobic manure lagoon Liquid 1968-70.

Number	Treatment	Applications	Amount In.	Tile drainage concentration, mg/l COD	Total-N	P	Cl^-
				513^b	335^b	72^b	110^b
1968-1	1.5 in. at 70% A.M.[a]	9	13.9	17	65	0.2	75
1968-2	3.0 in. at 70% A.M.[a]	7	20.9	30	52	0.7	80
1968-3	1.5 in. at 95% A.M.[a]	10	15.2	30	54	0.4	76
1968-4	3.0 in. at 95% A.M.[a]	11	30.5	32	72	0.7	79
Prec.			12.5				
				939^b	306^b	73^b	127^b
1969-1	2.0 in. at 750 mb[cd]	8	15.7	18	60	0.2	75
				22^d	9^d	0.3^d	21^d
1969-2	2.0 in. at 750 mb[c]	13	29.3	145	76	2.1	95
1969-3	2.0 in. at 750 mb[ce]	9	18.4	22	58	0.2	75
				43^e	296^e	1.4^e	21^e
1969-4	2.0 in. at 350 mb[c]	24	48.1	120	98	2.8	91
Prec.			31.1				
				1332^b	389^b	74^b	159^b
1970-1	2.0 in. weekly in 3, 2/3-in. daily applications	13	27.3	106	128	3.5	101
1970-2	2.0 in. at 750 mb[c]	7	15.3	147	117	3.4	119
1970-3	2.0 in. bi-weekly in 3, 2/3 in. daily applications	7	14.8	93	111	1.0	100
1970-4	2.0 in. at 350 mb[c]	15	31.4	280	149	7.0	132
Prec.			32.0				

[a] available moisture in soil at 6-in. deep, gravimetrically
[b] lagoon liquid caught on plot surface
[c] millibars soil suction at 6-in. deep
[d] tapwater after stored in reservoir and pumped to plots
[e] tapwater + N(NH_4NO_3 + Urea) after storage in reservoir and pumping to plots.

the results somewhat because it was assumed that all flow caused by irrigation was of uniform quality; however, later work showed that quality improves shortly after irrigation ceases. The low flow between events was not sampled. The porous cups and well points (2 in the plot and 3 pairs in adjacent unirrigated areas) were sampled as time permitted, usually every 2 to 3 wk.

RESULTS

Phase 1. Anaerobic Lagoon in the System. 1969

Manure transport:

During this phase each flush tank discharged 85 gal once each hour. Cleanliness of the pens varied according to position and time. Just after the flush the top two pens (15 ft of the flushing gutter) were essentially clear of all manure. A hydraulic jump occurred after this distance and cleaning was less spectacular. The action of the pigs' feet in the gutter was very beneficial since this kept the manure more mobile. Watching the slurry at the lower end of the house showed that it had adequate velocity to transport 1 in. dia x 2 in. manure pellets by rolling. At no time was old septic manure observed in the pens containing pigs, but such conditions would be observed if a pen were empty. The width of the gutter, 42 in., was too great for the available flow, hence some meandering would take place in empty pens, thus forming banks of decomposing manure. Since the majority of the pens were occupied all the time these manure banks were not important. An area in which settled manure solids caused difficulties was the 10 ft section of gutter between the last pen and the sewer. Fly breeding was observed here during the summer, but this was not difficult to control by periodic hosing down.

Buildup of manure solids near a wall has also been observed by Jones et al. (35) and they suggest that a shallow 6 in. wide trough be formed in the gutter at the wall side. This change in channel cross section increases velocity and shear, thus improving manure transport. The report by Jones et al. (35) also lends strength to the theory that flow rates in a long flushing gutter are determined more by the hydraulic characteristics of the channel than by the rate of release of water at the head.

Fecal parameters:

Samples were collected for analysis daily. The collection was made in the early afternoon immediately following a flush. The manure slurry sample was taken by dipping three 300 ml samples from an open portion of the sewer leading to the lagoon. The three subsamples were spaced equally during the time the wave of manure passed by the sampling point and were composited. The samples from the lagoon and the settling tank were taken from taps on the pump discharge lines. All samples were transferred to the laboratory for analysis within 10 min.

The difference between the raw manure and returned effluent strengths, correlated with the quantity of water discharged from the flush tanks and the pig population, was used to obtain fecal parameters for the individual pig. These are presented in Table 3 and it is seen that they agree well with values in the literature as summarized by Muehling (47). The volume of water contributed by waterer spill, urine and feces was unknown during this phase.

Table 3. Fecal production per pig. Phase 1

All in units of lb/100lb pig day

Month	Feb.	Mar.	Apr.	May	Avg.
COD	0.5	0.89	0.87	0.61	0.72
BOD_5 (Normal)	0.17	0.36	0.38	0.24	0.29
Total N	0.04	0.045	0.03	0.032	0.037
Volatile solids	0.44	0.5	0.36	0.34	0.41

Lagoon as a treatment system:

The lagoon is an unmixed anaerobic digester with an unusually long hydraulic detention time of about 200 days. Some mixing is observed during the warmer months of the year when boils of sludge lifted by CO_2 and CH_4 come to the surface periodically. During the winter it is postulated that the methane forming phase of bacteria are dormant because of the very low temperatures, and though the main function of the lagoon will be settling there should also be some waste liquefaction by facultative bacteria. O'Rourke (53) has found that at $59^{\circ}F$ ($15^{\circ}C$) methane formation was inhibited, even at solids retention times of 60 days.

Reference to Table 4 and Figure 12 shows that the COD in the lagoon supernatant was approximately constant from February to April when the temperature stayed around $34^{\circ}F$ (Figure 13). Vigorous biological activity and gas production were well underway by May as the temperature increased, and it is evident that the accumulation of winter COD is readily digested. Only one temperature measuring location was used, so this work is insufficient to contradict O'Rourke's (53) findings. However the present

Table 4. Loading and treatment efficiency Phase 1

	Values based on measured parameters				
	Lagoon				
Month	February	March	April	May	Avg.
Number of pigs	640	661	618	666	646
% COD removal	--	89.5	91	95.5	92.0
% BOD_5 removal	78	88	88	96.5	87.6
% total nitrogen removal	57.5	64	60.5	65	61.8
% volatile solids removal	89.5	91	91	86.5	89.5
VS loading, $lb/ft^3 day \times 10^3$	3.54	4.01	2.72	2.92	3.3
Hydraulic detention time, days	202	178	213	264	214
	Oxidation ditch				
% COD removal	70	76	64	--	70
% BOD_5 removal	84.5	81.5	83.5	--	83.2
% total nitrogen removal	26.5	6	27	6	16.4
% volatile solids removal	25	50	10.5	30	28.9
F/M ratio	0.058	0.039	0.028	0.006	0.033
Hydraulic detention time, days	13	11.5	14	17	13.9
	Overall				
% COD removal	95	97.5	97	87	94.1
% BOD_5 removal	96.5	98	98	87	94.9
% total nitrogen removal	66.5	66	69	67	67.1
% volatile solids removal	92	95.5	95	90.5	93.3

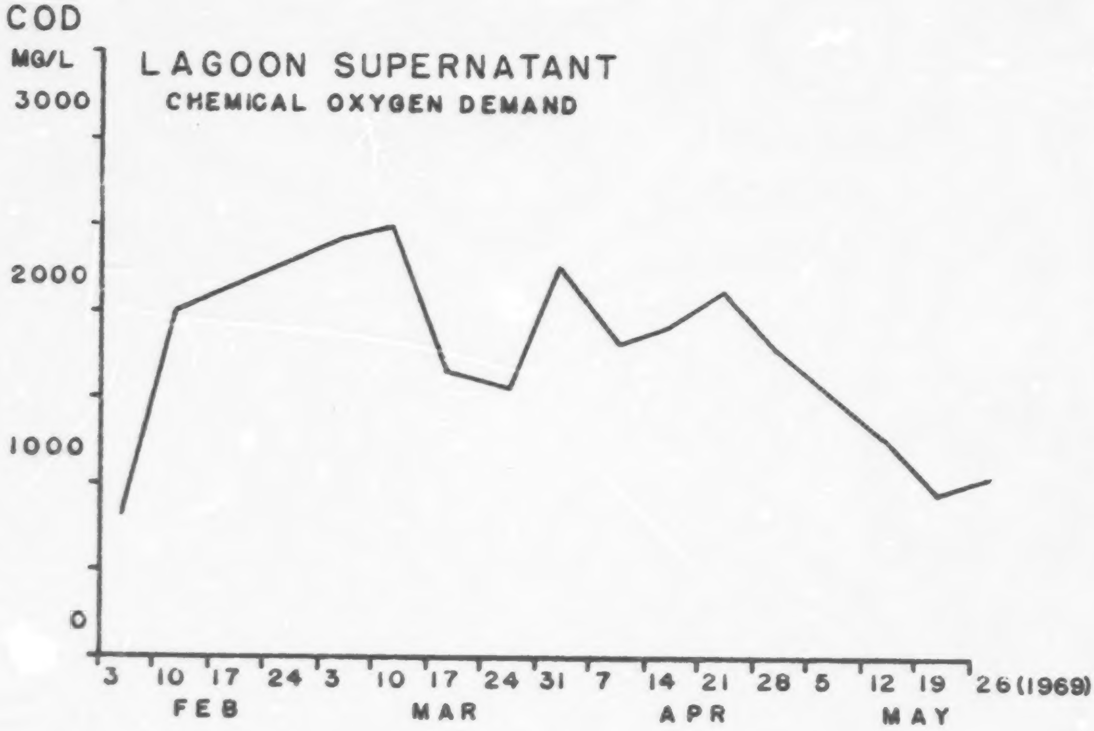


Figure 12. The chemical oxygen demand of the supernatant from the anaerobic lagoon during Phase 1

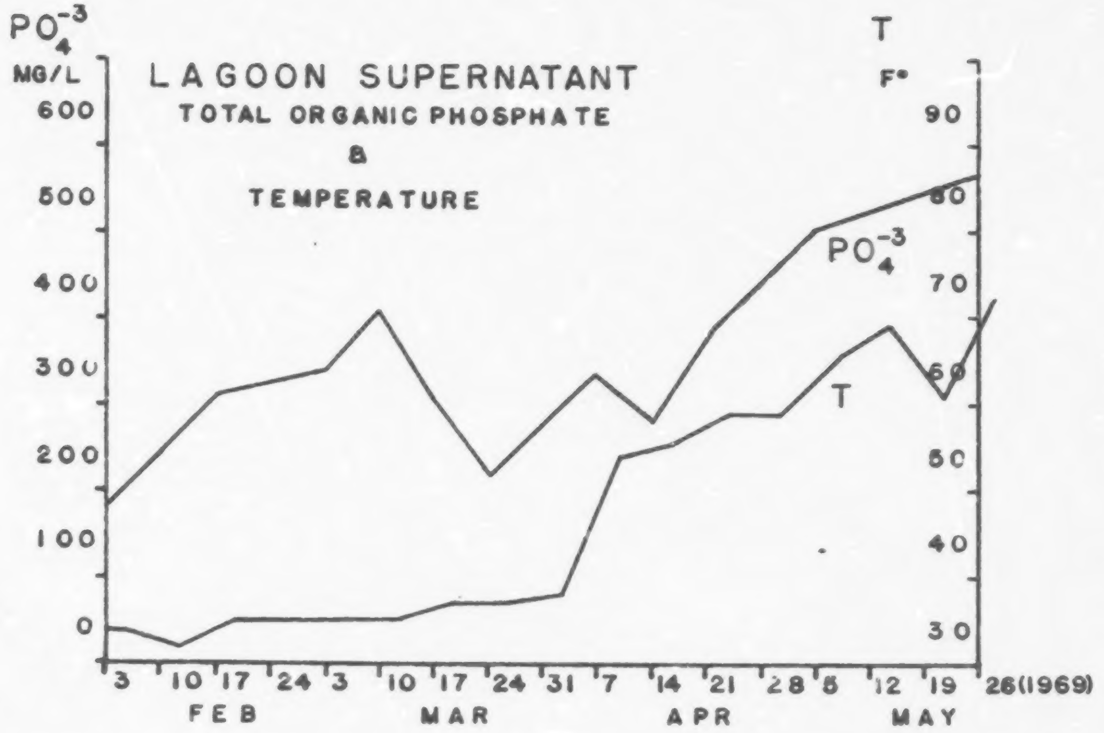


Figure 13. The total phosphate concentration and the temperature of the supernatant from the anaerobic lagoon during Phase 1

information does give some indication that methane formation (as indicated by gas production) does take place at 55°F. The lagoon is a good waste treatment system for livestock producers because it will accomplish large reductions in COD, BOD_5 and volatile solids; but it is very obvious, noting the values of the COD shown in Figure 12, that anaerobic lagoon effluent is not suitable for direct discharge into public waters.

Table 4 shows that there was an apparent removal of 60% in nitrogen on passage through the lagoon. Several mechanisms may be responsible: dilution, volatilization from the surface, cell formation and precipitation are but a few. Dilution cannot be neglected since it was found in Phase 2 that for each pig, about 1.26 gpd of extra liquid is added due to waterer leakage, washdown, urine, feces, etc. Additionally there will be unknown quantities of water added by runoff, snowmelt and ice melting. Volatilization is examined later in this report, but it is unlikely that loss by this mechanism is significant until the ice is melted. Cell formation in anaerobic systems is slow compared with aerobic systems, so, during the winter, the fraction converted to cell mass should be very small. Precipitation of nitrogen an inorganic form is a distinct possibility when magnesium, calcium and phosphate are also present. Recently Salutsky et al. (66) re-examined the formation of $Mg(NH_4)PO_4$ in a municipal anaerobic digester, because this product has potential value as a fertilizer and is also capable of reducing N and P in sewage plant effluents. The Handbook of Chemistry and Physics (82) shows that $Ca(NH_4)PO_4$ and $Mg(NH_4)PO_4$ are poorly soluble in cold water and neutral pH. No measurements of Ca or Mg were made but some estimates can be made. The feed used in Unit K is compounded according to Iowa State University recommendations found in Hodson et al. (28). The feed contains 0.66% of Ca on a dry weight basis, and a pig weighing 125 lb (mean weight of the pigs in the house) will consume 6.4 lb of feed each day. The Agricultural Research Council U.K. has surveyed the literature on swine nutrition up to 1967 and they conclude that Ca stored in the finishing pig body amounts to 7.5×10^{-3} lb/lb of daily gain. Hence, at a daily gain of 1.75 lb/day (Hodson et al. (28)) the retention rate of Ca will be about 1.3×10^{-2} lb/day. The amount of Ca fed is 4.26×10^{-2} lb/day; hence, an average 125 lb pig will excrete about 2.95×10^{-2} lb/day of Ca. Using values of 646 pigs in the house (Table 4) and 4000 gpd for the flush quantity, then the expected Ca concentration in the manure slurry is 570 mg/l. Observations indicate that P is not limiting; hence, if all the Ca fed remained in the lagoon in the form of insoluble $Ca(NH_4)PO_4$, then this compound could account for 200 mg/l of the influent N.

Figure 13 shows an increase in phosphate concentration after the spring thaw. At first this was attributed to organic material being lifted from the bottom and mixed in the main mass of liquid. However the analysis of the crystalline deposits found on the pump led to some other possibilities. As mentioned previously, it is known that Ca^{+2}, Mg^{+2}, NH_4^+ and PO_4^{-3} will form a variety of insoluble, or poorly soluble compounds. Under laboratory conditions, $Ca_3(PO_4)_2$ or $Ca(NH_4)PO_4$ (hydration status not known for either compound) are both very easily resuspended by mild agitation. The Murphy and Riley (48) method for phosphate determination uses an acidic solution of ammonium molybdate and ascorbic acid. Since $Mg(NH_4)PO_4$

is soluble in weak acid, it is assumed that this phosphate test will measure the phosphate in $Mg(NH_4)PO_4$, and most probably other calcium and magnesuim phosphates as well. Hence it is postulated that some part of the phosphorus increase observed during the spring biological activity is due to particulate inorganic phosphates being brought into suspension.

The total coliform reduction in the lagoon varied with time. During the period (up to the end of March) when the lagoon was quiescent, the percentage remaining in the lagoon effluent was 0.25%. As the biological activity increased the count varied widely and in May the least count was a percentage remaining of 0.053%, and the greatest 0.6%. This variation could be correlated with the suspended solids carried in the lagoon effluent.

The dissolved inorganic solids and chloride concentrations in the lagoon fluctuate according to rainfall. A discussion of these measurements in the later section on the results of Phase 2 explains more clearly why no upward trend in dissolved inorganic solids concentration was noted.

Oxidation ditch as a treatment system:

The literature (51, 67) indicates that treatment efficiency will primarily be a function of influent strength and MLSS content. The volume of the treatment tank will be a secondary variable related to the solids level that can be carried in accordance with the solid/liquid separation devices available. Another secondary variable is the aeration capacity. Because the system is operating in the extended aeration regime, aeration capacity may approach the COD plus nitrification requirements of the influent.

It should be noted that the ditch was not initially void of biological material because of its intermittent use after the preliminary stuay (70). Together Figures 14 and 15 show that there was little change in the BOD_5 of the settled effluent as the level of volatile suspended solids in the ditch increased. This seeming contradiction to McLellan and Busch (51) may not be so, since their theory relates to soluble BOD_5 and the supernatant BOD_5 measured was known to contain some fine suspended matter. Middlebrooks et al. (45) have also examined the composition of the soluble and suspended fractions from extended aeration and they conclude the major part of the BOD_5 is present in the solids.

Table 5 shows how the food to microorganism ratio varied through the test. Food is based on BOD_5 concentration and microorganisms are based on MLSS so that values of this parameter may be compared with those of Scheltinga (67). The wide variation in influent load to the ditch was caused by mechanical difficulties with the flush tanks. The F/M ratios are generally less than the 0.05 lb BOD_5/lb MLSS value recommended by Scheltinga (67). This indicates that the ditch was larger than necessary since no difficulties were encountered in settling, even at the highest MLSS encountered. The sludge volume index was generally less than 50 and the greatest value was only 66.

McCarty and Brodersen (49) have examined the extended aeration process and they conclude that a mathematical model may be written for a steady state condition. This model starts with the equation

$$\frac{dM}{dt} = a\frac{dL}{dt} - bM_a \qquad (2)$$

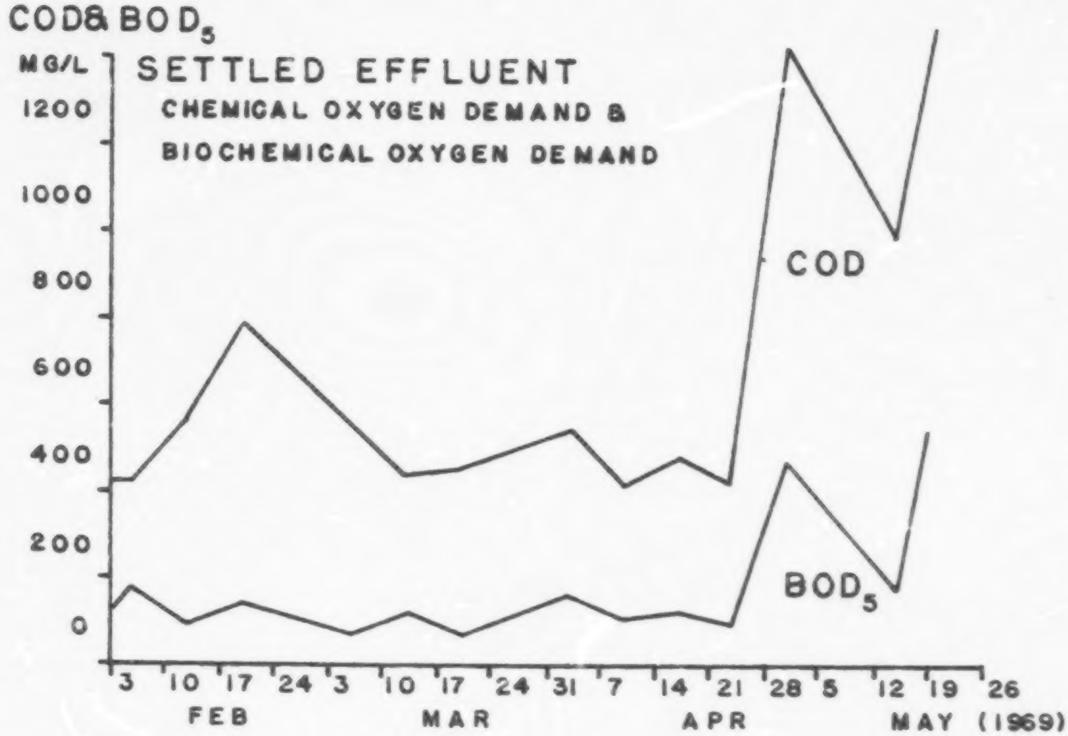


Figure 14. The chemical oxygen demand and the five day biochemical oxygen demand concentrations of the settled effluent from an oxidation ditch fed anaerobic lagoon supernatant during Phase 1

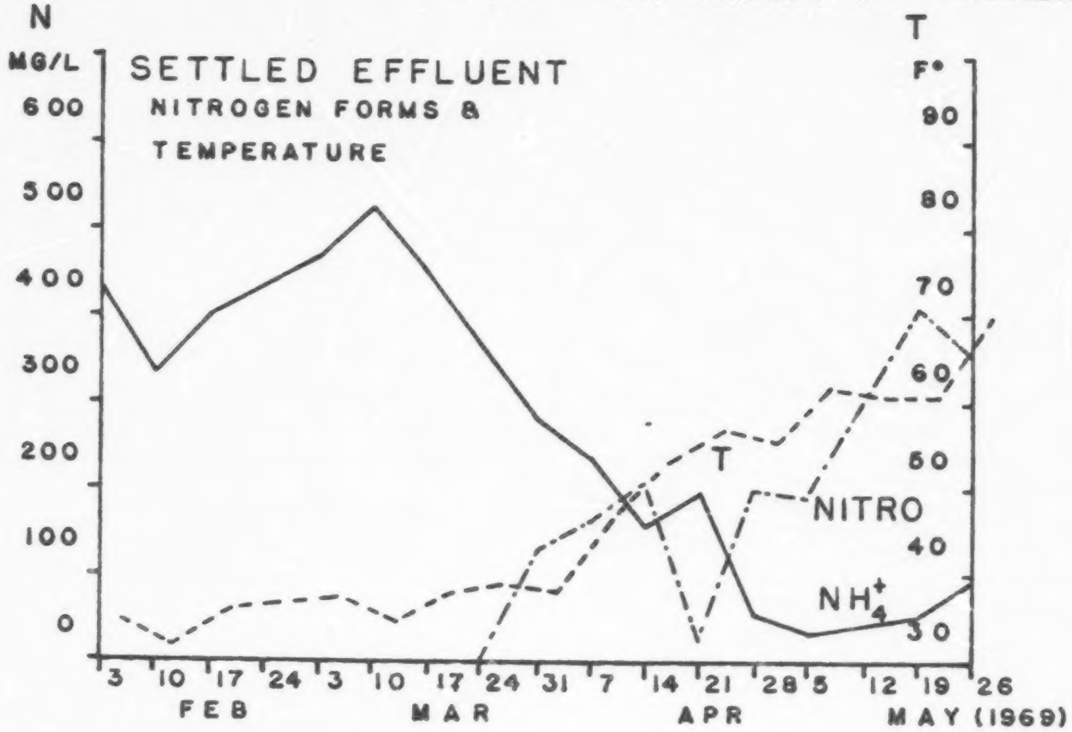


Figure 15. The forms of nitrogen and the temperature found in the settled effluent from an oxidation ditch fed anaerobic lagoon supernatant during Phase 1

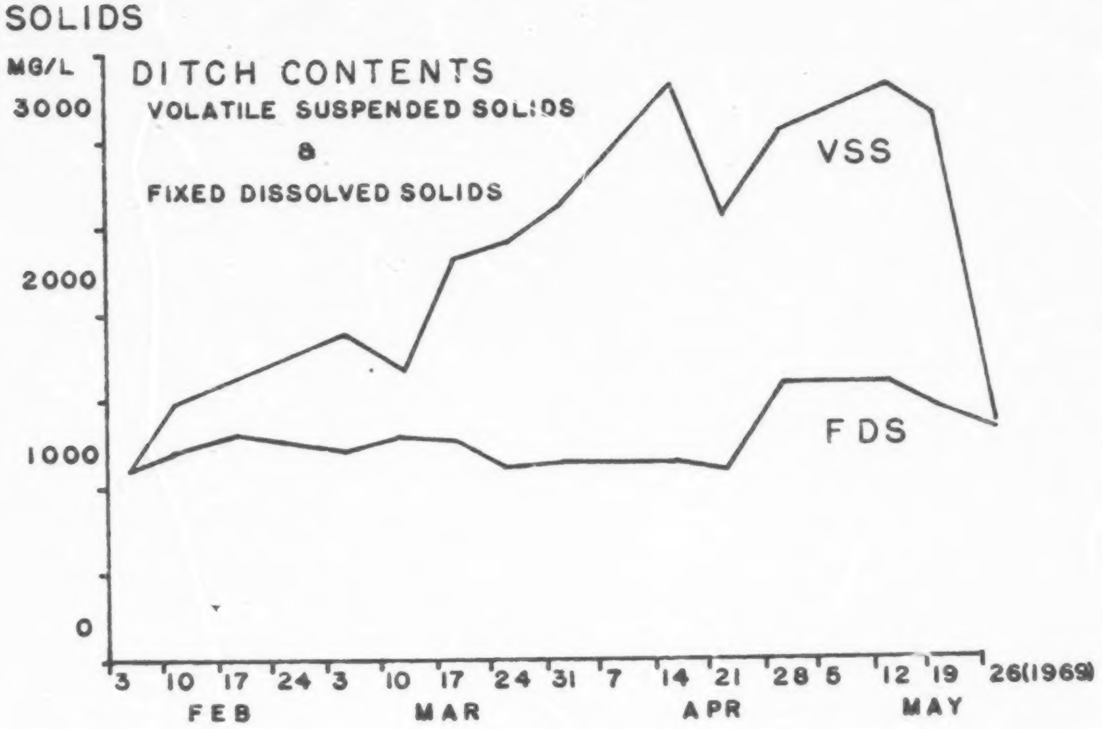


Figure 16. The volatile suspended solids and the fixed dissolved solids concentrations found in an oxidation ditch fed anaerobic lagoon supernatant during Phase 1

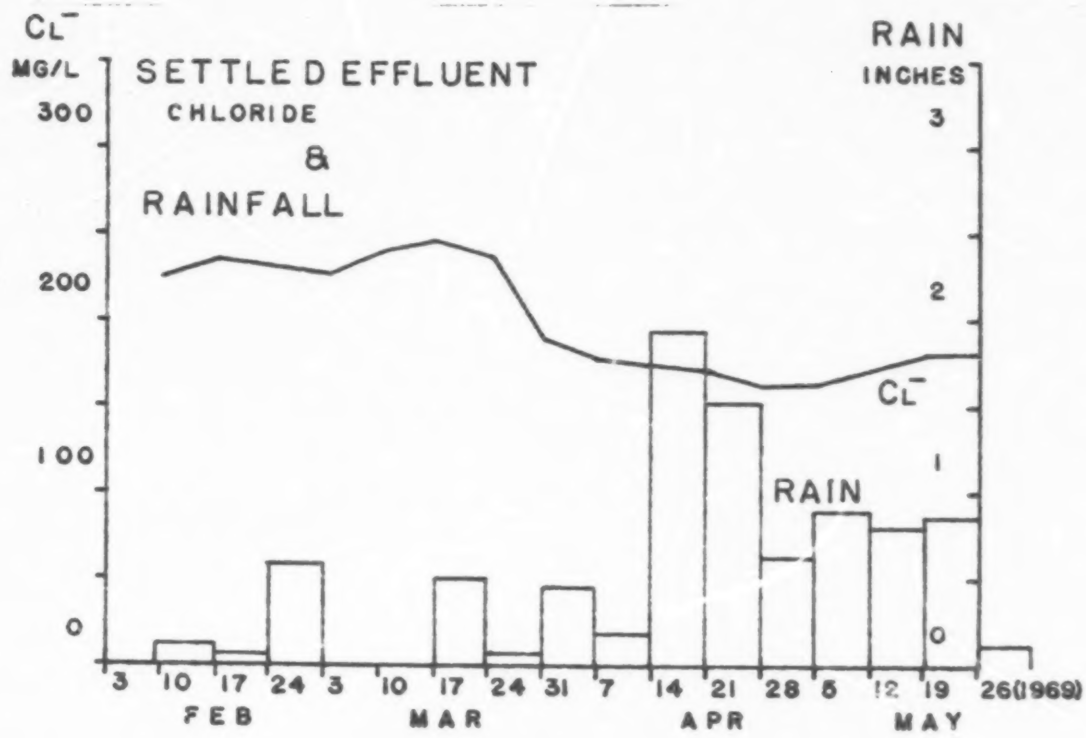


Figure 17. The chloride ion concentration found in the settled effluent from an oxidation ditch fed anaerobic lagoon supernatant and the rainfall record for a location near the lagoon during Phase I

Table 5. Food to microorganism ratio in the oxidation ditch. Phase 1

Date Day/Mo/Yr	BOD_5 lb/Day	MLSS lb	F/M $lbBOD_5$/ lbMLSS
3/2/69	19.5	498	0.0392
10/2/69	32.9	560	0.0588
17/2/69	49.4	1050	0.0470
24/2/69	--	--	--
3/3/69	33.5	954	0.0351
10/3/69	69.1	851	0.0812
17/3/69	25.7	1010	0.0254
24/3/69	27.6	1170	0.0236
31/3/69	30	1240	0.0242
7/4/69	--	--	--
14/4/69	42.6	1680	0.0254
21/4/69	40.6	1310	0.0310
28/4/69	31.7	1400	0.0226
5/5/69	--	--	--
12/5/69	6.98	1500	0.00465
19/5/69	13.8	1430	0.00965
26/5/69	8.53	902	0.00945

Where

$\frac{dM}{dt}$	Rate of generation of volatile suspended solids
a	Cell yield coefficient
$\frac{dL}{dt}$	Rate of soluble substrate removal
b	Endogeneous decay coefficient
M_a	Mass of organisms undergoing endogenous respiration

The results of this research were examined carefully to determine if a and b could be evaluated; but it was concluded that since the system never achieved steady state, no use could be made of the model. The model is considered good for the process in hand because it allows the prediction of the effluent BOD_5, incorporating all the solids discharged. It is interesting to note that McCarty and Brodersen (49) conclude, on theoretical grounds, that the effluent BOD_5 is a function of the suspended solids and that this was verified by Middlebrooks et al. (45) using chemical analysis.

The jump in COD and BOD_5 observed in late April was due to an unintended discharge of sludge from compartment A (Figure 4). Before this occurence the effluent from the settling tank was slightly turbid, odorless and pale yellow in color. Similar descriptions were made by Clayton and Feng (15) and Scheltinga (67).

The maximum aeration capacity of the rotor under standard conditions (68^oF, 0 mg/l DO and p_{atm} = 29.92 in Hg) at 7.5 in. immersion and 75 rpm was 102 lb O_2 per day. Reference to Table 5 shows that this was more than adequate to cope with the influent load. Such excess aeration capacity should have ensured nitrification but Figure 11 shows that this did not occur for 8 wk. The minimum solids retention time for nitrifying bacteria, given by Balakrishnan and Eckenfelder (7), is 3 days at 68°F (20°C); thus it is assumed that the delay was a temperature effect. Ludzack and Ettinger (43) have indicated that nitrification is insignificant below 41°F (5°C). Excess aeration capacity is evident from Figure 15 because the ammonia N recorded before nitrification is almost entirely replaced by oxidized N. The irregularity during the week beginning 21st April was due to a period when the rotor stopped.

Reduction in volatile solids was not as great as had been expected. This is attributed to fine suspended solids being carried over in the effluent.

Phosphorus reduction during passage through the ditch varied. During the period from 3rd February to 14th April, when the solids level in the ditch was rising, there was an appreciable phosphate removal. However the last part of the test period was marked by solids decline and the trend of phosphate reduction disappeared. Nesbitt (52) has reviewed phosphate removal by activated sludge, and he reports that the VSS from a treatment plant can contain between 2.6% to 7% P by weight (dry solids), depending upon operating conditions. The upper figure was for a well aerated sludge with an F/M ratio of 0.5 lb BOD/lb MLSS. A linear regression applied to the VSS results from 3rd February to 14th April (Figure 16) yields a rate of growth of 11.6 lb/day. An estimate of the VSS removed in the returned effluent yields 4.56 lb/day. Therefore, enough P must be supplied to produce a total of 16.2 lb/day of VSS. Using a mean flow rate through the ditch during this period of 3770 gpd shows that a VSS phosphorus content of 2.6% would require 41 mg/l PO_4^{-3} to be removed from the flow, but the upper value of 7.0% would require 111 mg/l. Table 6 shows that the mean difference between influent and effluent during the same time period was 118 mg/l. Even allowing for the variability in measurements, it does appear that the sludge was storing a rather high quantity of phosphorus, especially considering the F/M ratio was only 0.04 lb BOD_5/lb MLSS during this period, compared with the value of 0.5 lb BOD_5/lb MLSS quoted by Nesbitt (52).

For reasons discussed earlier, the chloride concentration shown in Figure 17 does not show the expected upward trend. The rapid decline in late March is attributed to the melting of the lagoon ice cover.

The record of pH in Table 6 may be correlated with the establishment of nitrification by the end of March. The early pH was around 8 when the ammonia level was high, but this dropped to 6.5 after nitrification was well established. This drop in pH is mentioned by Adema (1) and also by McCarty and Brodersen (49).

Power requirements:

The average power consumed by the rotor on a wire to water basis was 57.9 kwh/day. The whole test period yields an average value of BOD_5 destroyed in the ditch of 24.8 lb/day which gives a power demand, per unit of BOD_5 destroyed, of 2.3 kwh/lb BOD_5. Nitrification was extensive, suggesting that the aeration capacity of the rotor was in excess, so the value of 2.3 kwh/lb BOD_5 may be unrepresentatively high. On an individual pig basis, the power requirement is 0.09 kwh per pig.

Table 6. Phosphate and pH in the lagoon and returned effluent. Phase 1

Date	Acid hydrolysed phosphate			pH	
Day/Mo/Yr	Lagoon mg/l	Ret. Eff. mg/l	Diff. mg/l	Lagoon	Ret. Eff.
3/2/69	192	73	119	--	--
10/2/69	205	84	121	7.4	8.3
17/2/69	230	93	137	--	--
24/2/69	--	--	--	--	--
3/3/69	228	93	135	7.4	8.1
10/3/69	278	120	158	--	--
17/3/69	286	116	170	7.5	8.2
24/3/69	160	110	-50	7.5	8.2
31/3/69	214	96	118	7.4	8.0
7/4/69	200	80	120	7.4	7.9
14/4/69	280	128	152	7.6	7.8
21/4/69	350	154	--	7.5	7.7
28/4/69	324	784	--	7.3	6.3
5/5/69	530	660	--	7.3	5.6
12/5/69	--	--	--	--	--
19/5/69	428	616	--	7.3	6.2
26/5/69	440	468	--	7.3	6.8

Phase 2. Manure Diverted to the Ditch

Manure transport:

This phase started using a 1 hr flush period, but difficulty with rising sludge in the settling tank was experienced. The flush interval was increased to 2 hr in the hope of improving settling. The flush quantity was increased by raising the float valves in the tanks to compensate for the longer interval. In practice, flush quantities were variable, but the average flush was judged to be 100 gal for each tank. Liquid flow into the tanks was not measured accurately during this part of the test because the water meters were not reliable. The pen cleanliness did suffer from the reduced water use, but the effect was marginal.

Fecal parameters:

The values presented in Table 7 are lower than those shown in Table 3 (Phase 1). They are also lower than those reviewed by Muehling (47). The discrepancies are ascribed to faulty sampling technique. During Phase 1 the samples were collected after lunch when the pigs were fully active, whereas sampling in Phase 2 took place in the early morning, before the morning feed. The difference between the results clearly indicate the need of composite 24 hr sampling.

The non-nitrifying BOD_5 test (outlined in Appendix A) is of some significance, if BOD_5 is to be used as a design parameter. Young (88) has indicated that 4.33 g of O_2 will be required for each 1 g of N converted to HNO_3 (this value adjusts stoichiometric values for cell growth and O_2 contribution by the autotrophic nitrifiers). Therefore the BOD_5 measured

Table 7. Fecal production per pig. Phase 2

	All in units of lb/100lb pig day						
Month	Aug.	Sept.	Oct.	Jan.	Feb.	Mar.	Avg.
COD	0.25	0.51	0.27	0.39	0.29	0.26	0.33
BOD_5 (Normal)	0.12	0.23	0.16	0.14	0.19	0.22	0.18
BOD_5 (Non-nitrifying)	--	--	0.11	0.12	0.12	0.14	0.12
Total N	0.024	0.025	0.04	0.026	0.033	0.025	0.029
Volatile solids	0.28	0.19	0.17	0.24	0.23	0.24	0.23
Total PO_4^{-3}	0.041	0.028	0.018	0.019	0.0086	0.0047	0.02

on raw manure, using a seed containing active nutrifiers, may be expected to give values appreciably greater than the true carbonaceous BOD_5 values. From the nature of the recycling system, in Phase 2 the raw manure was continuously seeded in this fashion. During the period 11th January to 1st March, when the mean difference between the normal and nitrification inhibited BOD_5 was 3870 mg/l, the mean value of NH_3 - N was 455 mg/l yielding a theoretical nitrogeneous oxygen demand (NOD) of 1970 mg/l. Although this theoretical value was only half the NOD measured, the discrepancy may well be due to the easily hydrolyzed organic N in the manure not measured in the NH_3 - N test.

Overflow quantity:

A better water balance was obtained during the second phase because the treatment volumes were better defined. The supply to the tanks was measured and an overflow weir maintained the ditch level constant. Discharge over this weir represented excess water entering the system. The cover over the ditch was not watertight so an unknown quantity of water was contributed by rainfall and snowmelt. Rainfall between 11th January and 15th February was minimal, hence the overflow originating during this period may be considered as originating entirely from Unit K. The mean value of overflow during this period was 773 gal, or 1.26 gpd per pig. Taiganides (75) reported an average winter value of 0.71 gpd per pig in the same building. The difference may be due to increased waterer leakage and less floor evaporation.

Oxidation ditch as a treatment system:

As in Phase 1 the oxidation ditch was started with some initial biological solids. The break in data collection between October and December was caused by equipment malfunction. The helical rotor sludge pump proved inadequate and some time elapsed before it was replaced by a diaphragm pump. Concurrent with this delay was an electrical fault in the rotor control gear. The fault was finally traced to corrosion in a switch. Although these difficulties were corrected by late December, reliable results during early January were prevented by a severe blizzard at the turn of the year.

As explained earlier, the fecal parameters obtained during Phase 2 are not considered representative, therefore the influent load to the system is based on earlier results obtained by Taiganides and Hazen (76). These measurements were also made on Unit K, but 24 hr composite sampling was employed. The decrease in COD, VS and N are shown in Table 8. Removal has been defined as:

Removal = Influent - (Effluent + Storage)

where the storage term was calculated from:

Storage = Daily rate of parameter concentration increase x Ditch volume

Since the settling tank was ineffective during the last part of the research, the solids concentration in the ditch was not very different from the concentration in the overflow. Therefore the solids retention time was approximately equal to the hydraulic detention time and, using 773 gpd as the discharge from the system, a detention time of 61 days was obtained. This long detention time accounts for the large decreases in COD, VS and

Table 8. Loading and treatment efficiency. Phase 2. Influent data based on Taiganides and Hazen (76) data for the 100 lb pig
COD = 0.84 lb/day VS = 0.70 lb/day N = 0.06 lb/day

	11 Jan.	18 Jan.	25 Jan.	1 Feb.	8 Feb.	15 Feb.	22 Feb.	Avg.
Number of pigs	582	616	632	621	596	613	626	612
Overflow rate, gpd	685	573	722	1160	725	2140	650	950
Hydraulic detention time, days	69	82	65	41	65	22	72	59
Chemical oxygen demand								
Influent, lb/day	611	647	664	652	626	644	657	643
Effluent, lb/day	56	57.4	112	144	65.3	314	63.5	
Storage, lb/day	151	182	73	-16.8	38.8	-42	-78	
Removal, lb/day	404	408	479	525	522	372	672	483
Percent removal	66.1	63	72.1	80.5	83.4	57.8	102.2	75.0
Volatile solids								
Influent, lb/day	509	539	553	543	522	536	548	538
Effluent, lb/day	46.5	18.3	123	118	64.1	228	57.8	
Storage, lb/day	22.4	69.8	63.9	-78.5	27.5	22.7	-78.8	
Removal, lb/day	440	451	366	504	430	285	569	435
Percent removal	86.4	83.7	66.2	92.8	82.4	53.2	103.8	81.2
Nitrogen								
Influent, lb/day	43.7	46.2	47.4	46.6	44.7	46	47	45.9
Effluent, lb/day	5.43	3.72	6.27	14.2	5.02	30.7	2.92	
Storage, lb/day	-10.7	6.16	6.43	-1.96	1.69	-7.02	3.37	
Removal, lb/day	49	36.3	34.7	34.4	38	22.3	40.7	36.5
Percent removal	112	78.6	73.2	73.7	85	48.5	86.6	79.7

Table 9. Food to microorganism ratio in the oxidation ditch. Phase 2. Based on 0.38 lb BOD_5/100 lb pig, Taiganides and Hazen (76)

Date Day/Mo/Yr	BOD_5 lb/Day	MLSS lb	F/M lbBOD_5/ lbMLSS
24/8/70	273	2275	0.12
31/8/70	273	3037	0.09
7/8/70	236	2966	0.078
14/8/70	236	--	--
21/8/70	236	2385	0.099
28/8/70	236	2581	0.092
5/10/70	219	2993	0.073
12/10/70	222	3558	0.062
28/12/70	273	4520	0.060
4/1/71	283	--	--
11/1/71	276	4810	0.057
18/1/71	293	4845	0.060
25/1/71	300	6156	0.049
1/2/71	295	6277	0.047
8/2/71	283	5186	0.055
15/2/71	291	6771	0.043
22/2/71	297	5932	0.050
1/3/71	314	5908	0.053

N noted in Table 8.

Because of the large increase in loading on the ditch, the F/M ratio is considerably different from Phase 1, as a comparison of Tables 5 and 9 shows. The F/M ratio becomes stable around 0.05 lb BOD_5/lb SS when the suspended solids level off at about 14,000 mg/1 (Figure 20). This value is at the upper end of the recommended range of solids concentrations given by Scheltinga (67).

Conventional waste treatment incorporates liquid/solid separation hence treatment efficiency is usually related to the oxygen demanding substances remaining in soluble or colloidal form in the effluent. When such a standard is applied to the returned effluent, both the BOD_5 and COD of the liquid portion of the effluent are quite low (Figure 18), indicating a high level of treatment by the system. However, in Phase 2, settling proved difficult even though the results obtained in Phase 1, where sludge densities of 20,000 mg/1 were found, indicated that settling should be possible at MLSS of 14,000 mg/1. Part of the settling problem is ascribed to the hair and husk now present in the influent. Denitrification also contributed to settling difficulties.

The solids generated in the ditch will need further treatment, since they are not stable and the mixed liquor putrefies readily on standing for more than 8 hr. The high level of sulphates (Figure 21), which are thought to be an unidentified component of the feed, are a source of odor, since these are reduced by a period of anaerobiosis. This sulphur transformation might prove to be a major disadvantage of recycling since the sulphate levels at the end of the test had exceeded 500 mg/1.

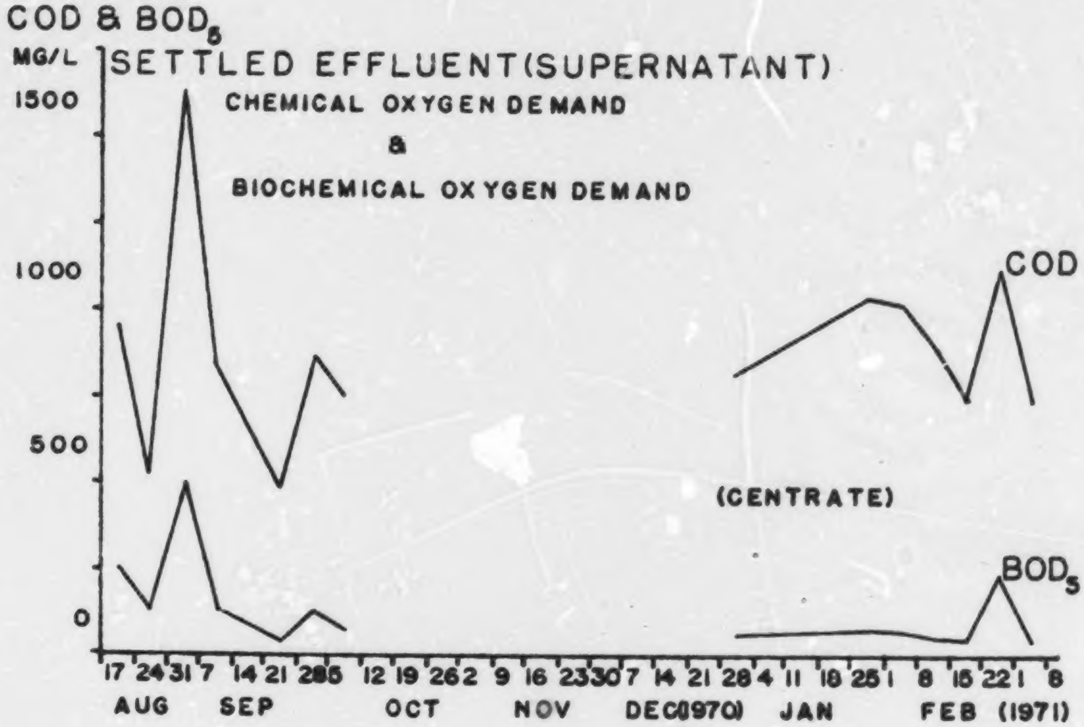


Figure 18. The chemical oxygen demand and the five day biochemical oxygen demand of settled effluent from an oxidation ditch fed raw manure. The results for August and September 1970 are for the supernatant; the results for January and February 1971 are for centrifuge supernatant. Phase 2

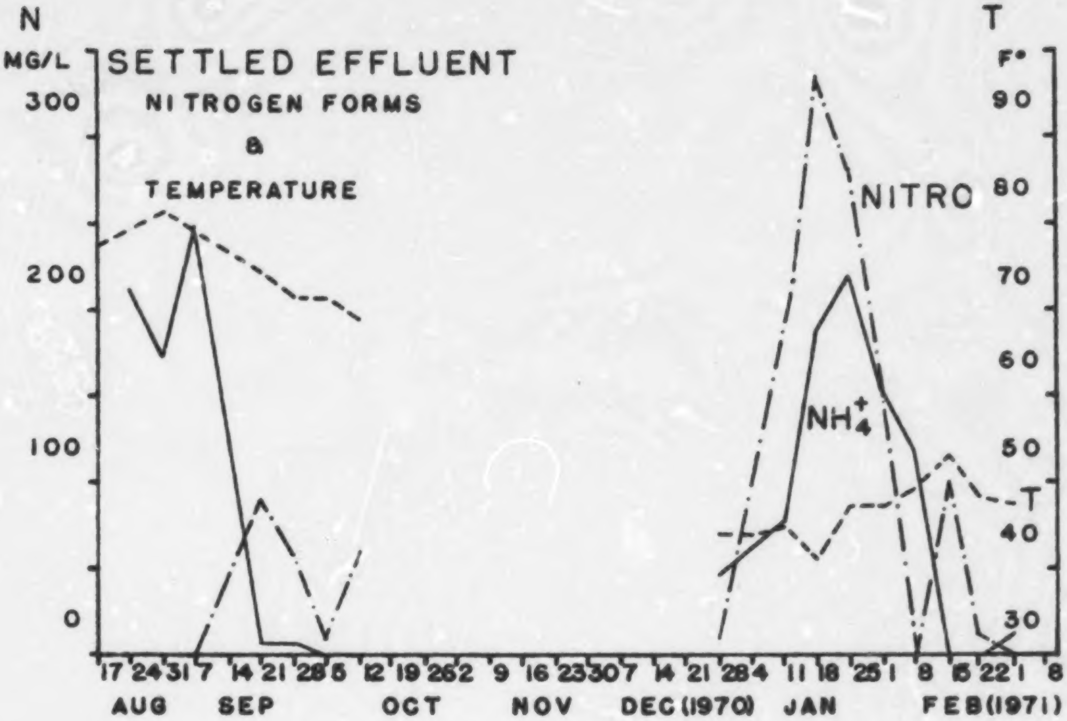


Figure 19. The forms of nitrogen and the temperature found in the settled effluent from an oxidation ditch fed raw manure; the results are for Phase 2

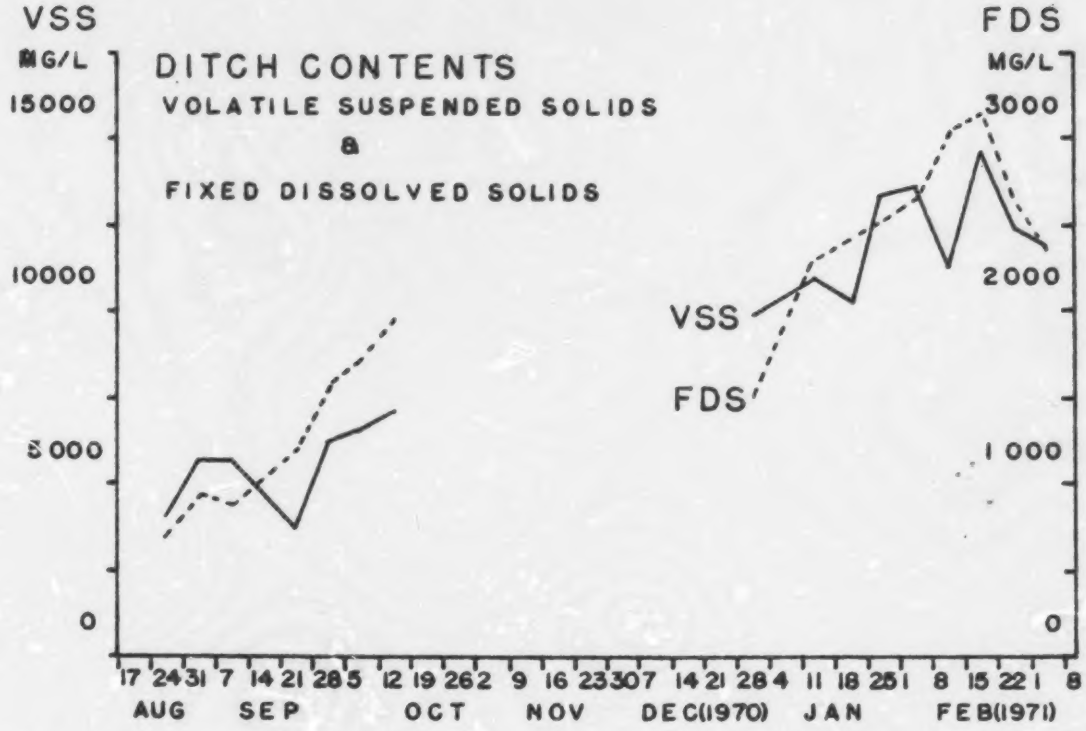


Figure 20. The volatile suspended solids and the fixed dissolved solids concentrations found in an oxidation ditch fed raw manure; the results are for Phase 2

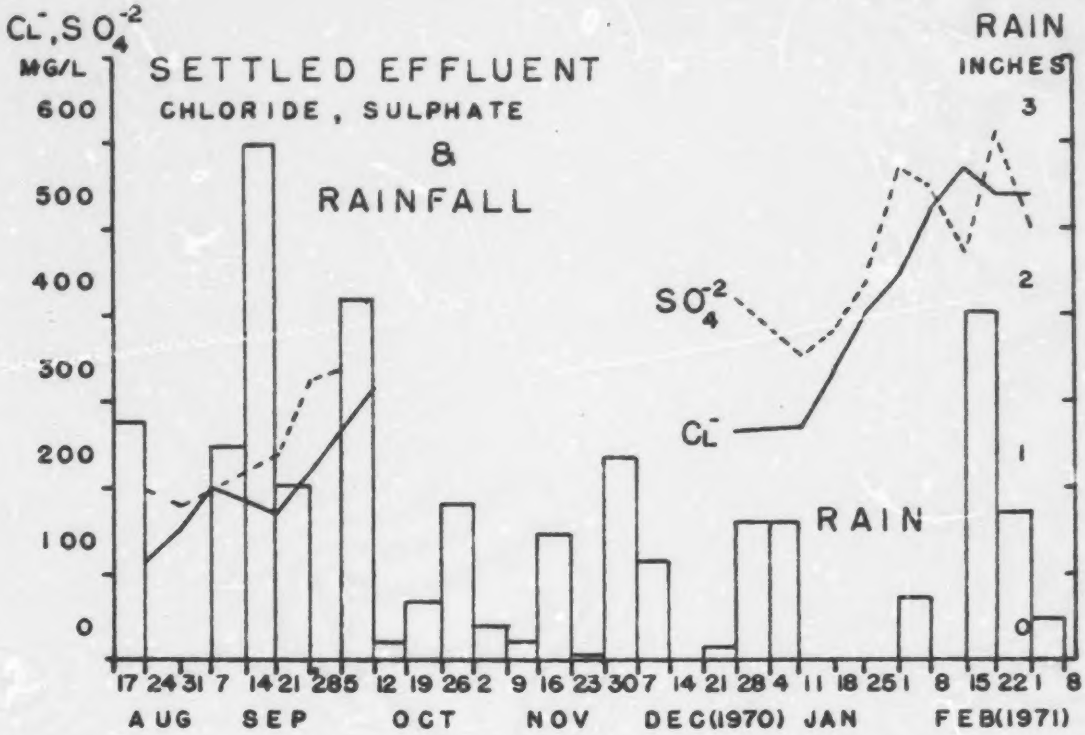


Figure 21. The chloride ion and sulphate ion concentrations found in the settled effluent from an oxidation ditch fed raw manure and the rainfall record in a location near to the ditch; the results are for Phase 2

Nitrogen removal in the ditch during Phase 2 was around 80% (Table 8) compared with 16% in Phase 1. This difference seems best explained by inspection of the aeration capacity available. The rotor was running at 12 in. immersion and 120 rpm, and Knight's (37) data shows that the aeration capacity was 504 lb O_2 per day under standard conditions (68^{o}F. 0 mg/l DO and p_{atm}=29.92 in Hg). The COD reduction amounted to a mean value of 484 lb/day; hence this would indicate that aeration capacity was very marginal, particularly since Loehr et al. (42) have shown that the oxygen transfer coefficient drops appreciably as the concentration of solids increases. Thus it is hypothesized that there is adequate dissolved oxygen in the zone just downstream of the rotor for nitrification to take place; but farther along the ditch oxygen is depleted by carbonaceous oxygen demand, thus denitrification takes place. Balakrishnan and Eckenfelder (6, 7) have studied both nitrification and denitrification and their findings are:

a) Nitrification.
   Solids retention time is to exceed 3 days at 68^{o}F.
   F/M ratio should be less than 0.8 lb BOD_5/lb MLSS.
   Nitrification increases as the ratio C/N increases.
b) Denitrification.
   MLSS level should be high.
   Readily available carbon source is necessary.
   Anaerobic conditions must prevail.
   High levels of nitrate nitrogen enhance denitrification.

All these conditions can be met as the liquor passes round the ditch. A qualitative indication of denitrification was the permanent formation of a scum layer around three quarters of the ditch length. Denitrification also took place in the settling tank since the levels of nitrate nitrogen measured in the returned effluent were consistently lower than those recorded in the ditch. Figure 19 shows that reduction of ammonia N did not correspond to increase in oxidized N, as in Phase 1.

The peak in both NH_3 - N and oxidized N observed in January, is most probably from hydrolysis of organic N accumulated in the ditch during the prolonged rotor stoppage in December. Phosphate results were very inconsistent. The difference between PO_4^{-3} contents of the influence and overflow effluent cannot be calculated because PO_4^{-3} levels were not measured for the overflow. However at termination of recycling the total PO_4^{-3} level was in excess of 1700 mg/l. Such a high level should have some potential as fertilizer.

Total coliforms were reduced by 93.5% during passage through the ditch (11th January to 1st March). This reduction exceeds that found in Phase 1, 85.9%, by a considerable margin. The improvement is a little surprising, because the concentration of suspended solids in the returned effluent during Phase 2 far surpassed that found in Phase 1. These total coliform counts are of rather dubious sanitary significance and in future both total and fecal counts should be made.

The rate of buildup of inorganic dissolved salts (Figure 20) has been suggested as being hazardous to biological treatment (Smith and Jenkins (69)). In fact, this is unlikely to be the case for recycling biological treatment systems where there is appreciable fresh water input from waterer spill and washdown. The concentration increase rate of an inert material may be analyzed mathematically if some reasonable assumptions are made. These are: that the flow-through rate of excess water is specified, that the rate of addition of the inert substance is specified and that the treatment system is completely mixed. Using the diagram and symbols shown on Figure 22 an equation relating concentration with time may be derived.

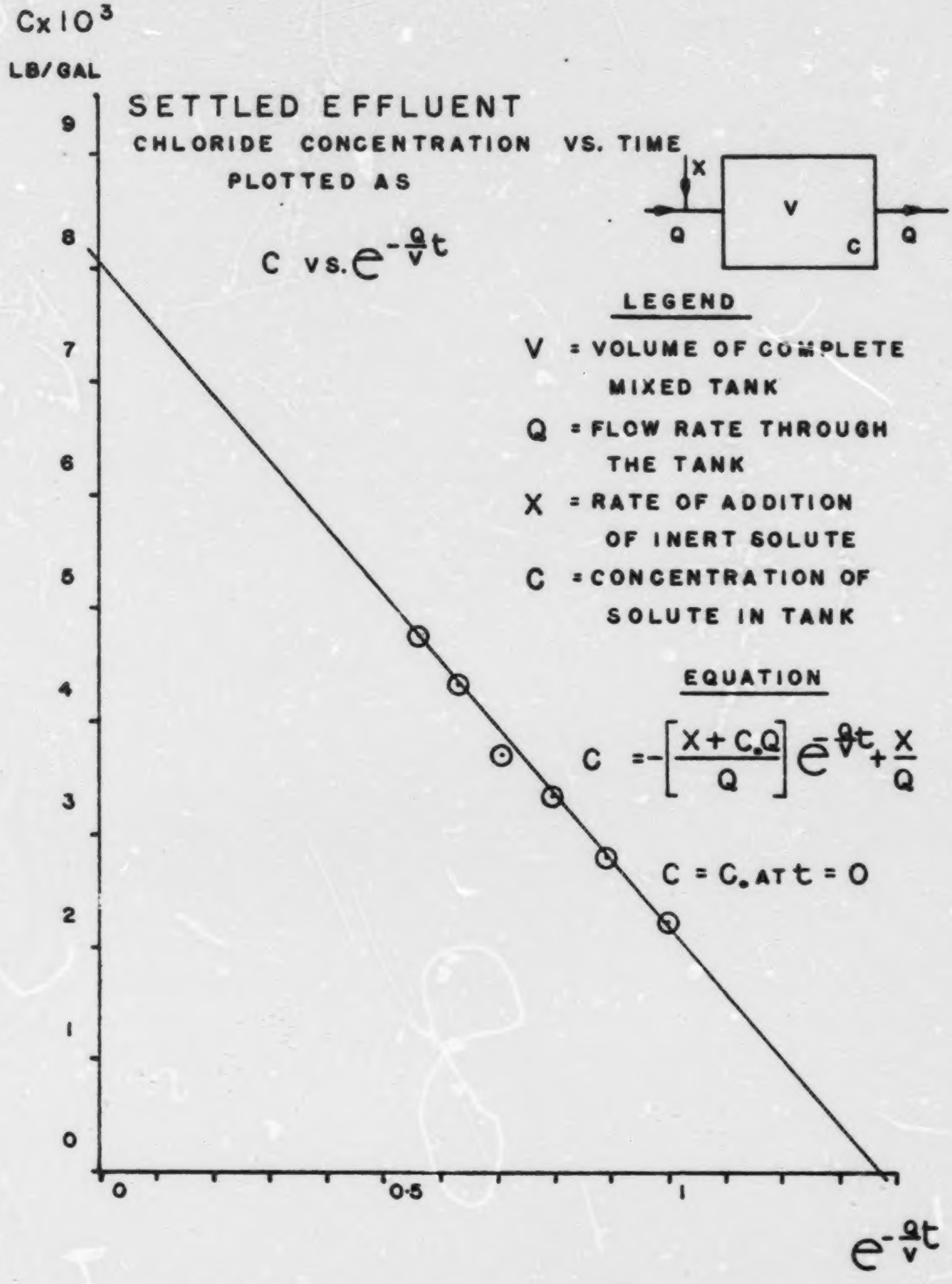


Figure 22. Chloride ion concentration and time for the settled effluent from an oxidation ditch fed raw manure; the results are for Phase 2

$$C = -\left[\frac{X + C_o Q}{Q}\right] e^{-\frac{Q}{V}t} + \frac{X}{Q} \qquad (3)$$

If $t = 0$ at $C = C_o$ and Q and V are both specified, the equation may be plotted in the linear form of C vs. $e^{-\frac{Q}{V}t}$. This has been done for chloride ion concentration in Figure 22, using the time span from 11th January to 15th February. This period was chosen because there was no significant rainfall (Figure 21) and the pig population was fairly constant. The values used in coding the data were $Q = 773$ gpd, $V = 47{,}000$ gal and $C = 2.25 \times 10^{-3}$ lb/gal (267 mg/l). The results show good linearity and yield a rate of Cl addition of 6.25 lb/day. This value may be compared with 11.5 lb/day of Cl in the pig feed and 7.5 lb/day from fecal measurements. If an arbitrary figure of 10 lb/day is chosen, then the ultimate concentration as $t \rightarrow \infty$ is $\frac{X}{Q}$, or 1550 mg/l. Using Equation 3 it may also be shown that it would take 140 days to reach 90% of this value. This is a low rate of increase, and Kincannon and Gaudy (36) could find no inhibition of continuous heterogeneous cultures even when the Cl concentration was changed from 0 to 30,000 mg/l in 24 hrs. It is felt that this analysis shows clearly why the expected trends in Cl increase were not observed in Phase 1. The total volume of the system (V) was about 10^6 gal, and this overshadowed the numerator of the exponent (Qt) in the period available. Moreover rainfall and snowmelt over the large area of the lagoon invalidate the assumption that Q was constant. Thus, in most biological treatment systems employing recycling, a combination of waterer spill and washwater, with realistic treatment volumes, will preclude salt toxicity. An exception might be for poultry; but even in this case, the results of Smith and Jenkins (69) suggest that salt concentrations can be very high before inhibition occurs.

Power requirements:

The average power consumed by the rotor during this phase was 146 kwh/day. Further calculation shows the power requirement per unit of BOD_5 destroyed to be 0.705 kwh/lb BOD_5. This value is nearly one third that derived in Phase 1 but in Phase 2 aeration capacity was marginal. The Water Pollution Control Federation (81) reports an average value of 0.446 kwh/lb BOD_5 for municipal plants employing surface aeration. A direct comparison between municipal practice and this study is not wise because of the much greater MLSS level here present. The individual pig power consumption amounted to 0.24 kwh/day per pig. Power consumption for the pumps was 20.4 kwh/day.

Aerobic Sludge Digestion

Objectives:

At the end of Phase 2, the ditch contained a fairly concentrated mass of unsettlable sludge. Jaworski et al. (31) have reported that extended aeration of activated sludge without further feeding can improve settling characteristics. The concept of aerobic sludge digestion has been applied to small municipal plants with some success (Ritter (64)), since it can achieve reasonable reductions in volatile solids and also produce a stable, easily-dried sludge.

The feed to the ditch was closed and contents were allowed to digest aerobically. Analyses were limited to solids, COD in mixed liquor and centrate, Kjeldahl nitrogen in mixed liquor and centrate, chloride and temperature. These analyses were chosen so that some measure of the volatile solids reduction could be obtained, and also some measure of the change in sludge composition with age in respect to VSS:COD:N ratios. The chloride concentration was measured so that an estimate could be made of rainfall dilution during the experimental period. Settleability was measured using the sludge volume index. One set of data points was obtained each week for 10 wk, the test was ended prematurely by incipient failure in a rotor countershaft bearing.

Analysis of results:

Spring 1971 was rather exceptional in Iowa, negligible rainfall occurring during the experimental period. The temperature varied from 47° to 67°F, with a mean temperature of 56.3°F. The results from Phase 2 had shown that the contribution of the centrate to the total volatile solids, COD or N was fairly small; hence attention was focused on the reduction of volatile matter, COD and N, in the suspended solids.

A graph of VSS vs. time is shown in Figure 23. The graphs of COD and N in the SS, being very similar, are not shown. All results show a jump between the 28th and 35th day. It is suspected that this jump may be due to resuspension of materials from the floor of the ditch.

BOD is often approximated by a first order curve

$$\frac{dL}{dt} = -kL \qquad (4)$$

and it seemed reasonable to examine the data from this aspect. The choice of first order curves by sanitary engineers is widespread. Even if it

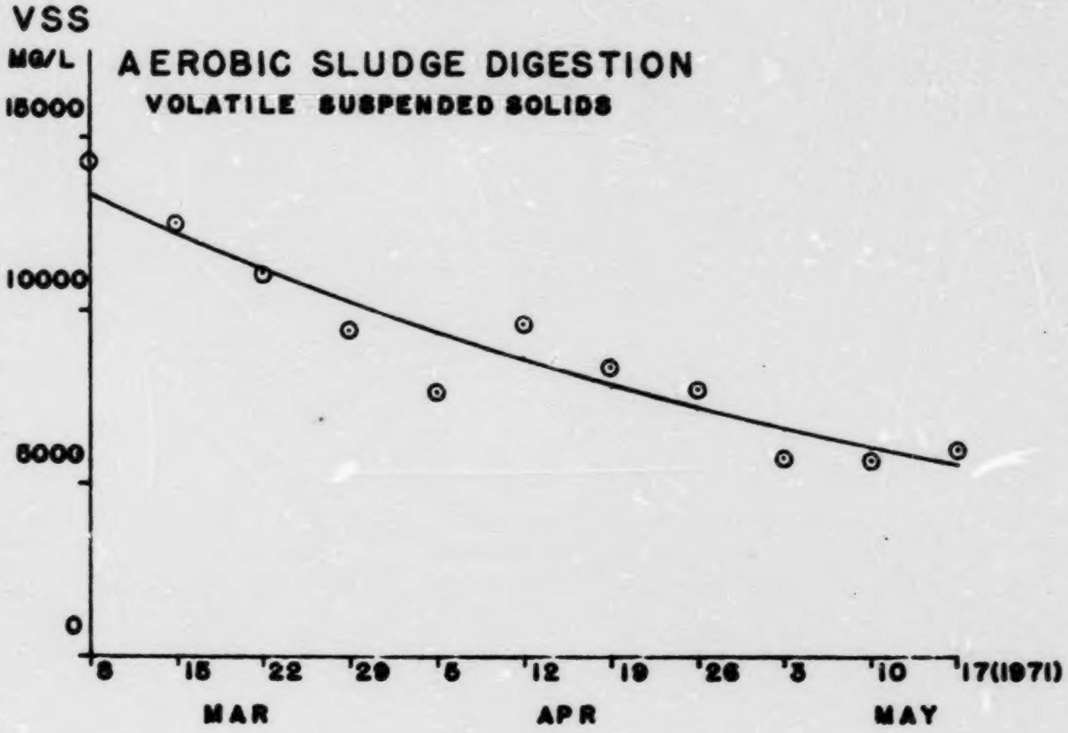


Figure 23. Reduction in volatile suspended solids during aerobic sludge digestion. After Phase 2

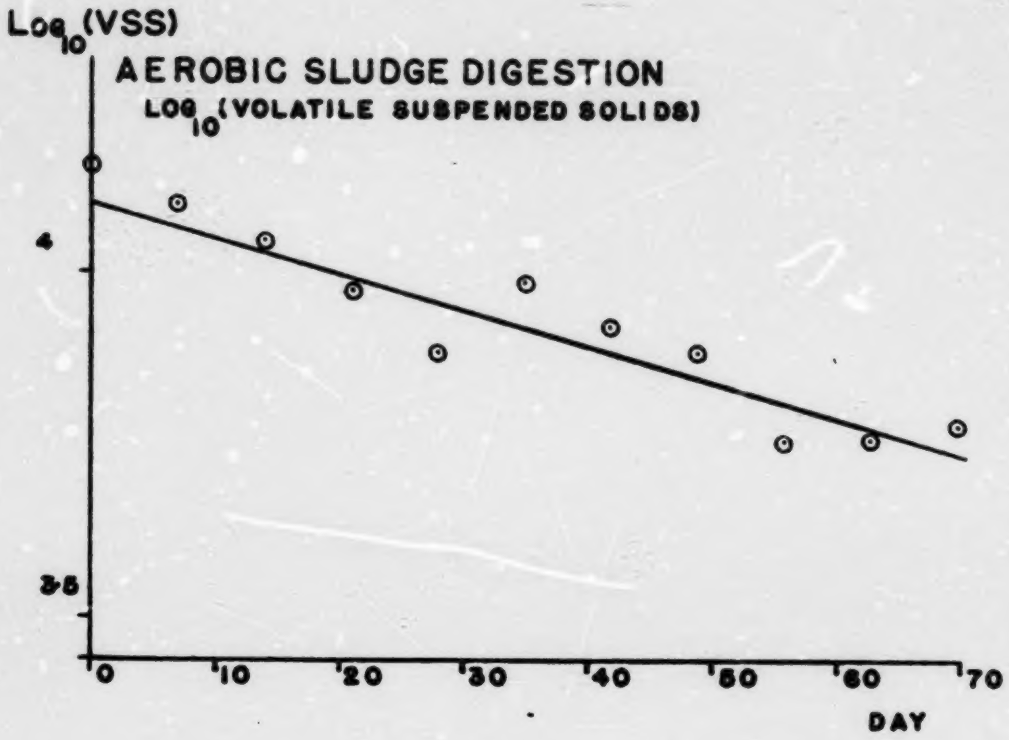


Figure 24. Logarithmic plot of the reduction in volatile suspended solids during aerobic sludge digestion. After Phase 2

has no easily derived theoretical justification, it does have very real practical value; since it allows experimental data to be codified for design purposes. The model chosen is

$$\frac{L}{L_o} = 10^{-Kt} \tag{5}$$

where

L	Parameter remaining after time t
L_o	Initial value of L at $t = 0$
K	Rate constant
t	Time in days

This equation may be rewritten as

$$\log_{10} L = -Kt + \log_{10} L_0 \tag{6}$$

and this should yield a straight line if a graph of $\log_{10} L$ vs. t is plotted. The value of L_0 and K have been obtained using linear regression (Ostle (54)). The graph of $\log_{10}$(VSS) vs. t is shown in Figure 24. The values obtained ignored the jump in parameters appearing after the 28th day.

Table 10. L_0 and K at a mean temperature of 56.3°F

Parameter	L_0 mg/1	K day^{-1}
Volatile suspended solids	13,350	0.00556
COD in SS	20,700	0.00583
Kjeldahl N in SS	1,390	0.00774

Tables 10 and 11 suggest that there is a slight change in sludge composition as the test progressed. The ratio of VSS:COD remained approximately constant as borne out by the similarity of the two K rates. However, the N content appears to decrease faster and shows an accordingly higher K rate.

Table 11. Ratios of VSS:COD:N in the suspended solids. Aerobic digestion

Date Day/Mo/Yr	VSS:COD:N		
8/3/71	1	1.57	0.0996
15/3/71	1	1.44	0.106
22/3/71	1	1.67	0.104
29/3/71	1	1.4	0.0943
5/4/71	1	1.59	0.0917
12/4/71	1	1.44	0.0784
19/4/71	1	1.89	0.0821
26/4/71	1	1.34	0.0741
3/5/71	1	1.61	0.085
10/5/71	1	1.33	0.0802
17/5/71	1	1.52	0.0724

A finding of this study is that aerobic sludge digestion does help settleability considerably. After 21 days, the SVI was 72 and the MLSS was still at 13, 800 mg/l (Table 12). It is surmised that the rather marginal aeration capacity during Phase 2 may have caused the bacteria to store substrate as increased polysaccharides in the slime layer and this layer prevented proper flocculation. The slime layer was digested by endogenous respiration once the influent was shut off. The rise in SVI noted in the last week of the test is not understood. The fraction of the sludge which was volatile did not change appreciably through the test, a similar finding to that of Randall and Koch (60).

Table 12. Suspended solids and SVI. Aerobic digestion

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l	Percent volatile	SVI ml/g
8/3/71	17,410	14,360	3,050	82.5	NS[a]
15/3/71	15,360	12,530	2,830	81.6	NS
22/3/71	13,760	11,080	2,680	80.5	72
29/3/71	11,820	9,410	2,410	79.6	NS
5/4/71	9,050	7,630	1,420	84.3	109
12/4/71	11,880	9,640	2,240	81.1	78
19/4/71	10,120	8,340	1,780	82.4	65
26/4/71	9,640	7,680	1,960	79.7	51
3/5/71	7,320	5,690	1,630	77.7	74
10/5/71	7,330	5,720	1,610	78.0	72
17/5/71	7,740	5,970	1,770	77.1	114

[a] NS = No settling.

Table 13 shows that the decrease in suspended solids COD was not accompanied by an increase in centrate COD. This seems to suggest this fraction of the COD was leaving as CO_2 instead of being converted to biologically refractile liquor.

Table 13. COD in centrate and suspended solids. Aerobic digestion

Date Day/Mo/Yr	Centrate COD mg/l	SS COD mg/l
8/3/71	1,240	22,600
15/3/71	1,060	18,040
22/3/71	915	18,490
29/3/71	856	13,140
5/4/71	760	12,140
12/4/71	703	13,910
19/4/71	657	15,740
26/4/71	581	10,320
3/5/71	547	9,150
10/5/71	440	7,630
17/5/71	406	9,090

In spite of the long detention time, the ditch liquor was not stable at the end of the test. A bottle of mixed liquor, held at room temperature (75°F), showed sludge rising and gave off a strong odor upon standing for 24 hr. This was unexpected, as Jaworski et al. (31) had indicated that a

60 day sludge was quite stable and did not putrefy on standing.

Application:

The data are not directly applicable since any process for animal wastes must be continuous. However, the beneficial reductions in VSS, COD and N, combined with improved settleability, suggest that extended aeration of the effluent from a first stage aeration process might occasionally be useful. The process could use a floating aerator and could be used for conditioning and storing excess sludge before final disposal by irrigation. The feasibility would hinge upon a livestock producer's location since the process is expensive both in terms of the initial capital outlay and power costs for operation. In general an anaerobic lagoon for storage and some treatment of excess water would seem more practical.

Pigs' Water Supply. Phase 2 Treatment System

Establishment of feeding trial:

All pigs in Unit K have access to the manure and flushing water in the gutter at the end of the pens. An experiment was conducted to determine how pigs would be affected if they were fed normally, but were forced to use the flushing gutter as their only drinking source. A group of six pens were established at the lower end of the flushing gutter and these were equipped with self-feeders. Ten pigs of similar age were placed in each pen on 9th December 1970 but the trial could not begin until 5th January 1971 because recycling of effluent had been interrupted by mechanical difficulties. The layout of the pens and how they were supplied with water is shown in Figure 25.*

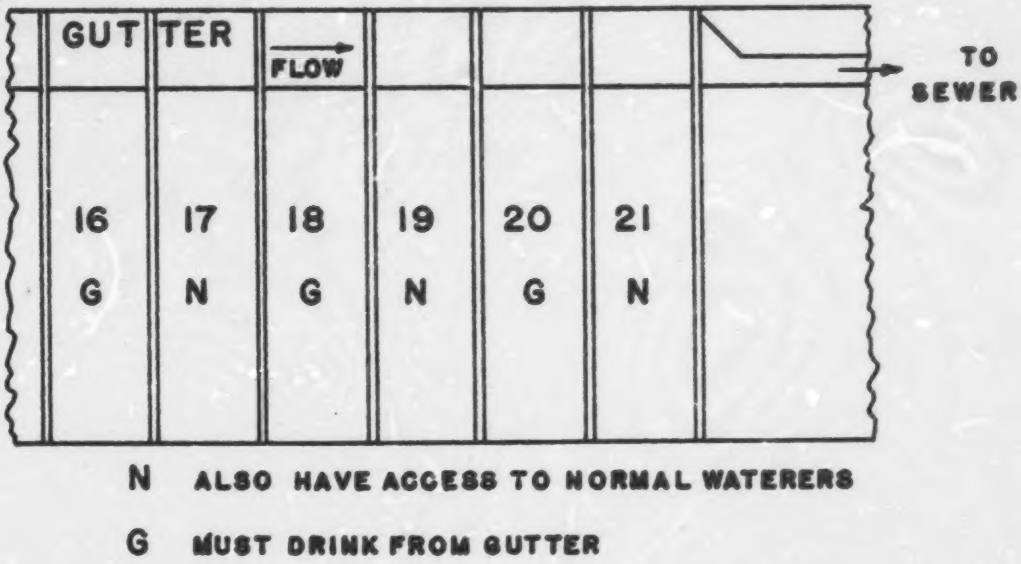

GUT TER
FLOW
TO SEWER
16
G
17
N
18
G
19
N
20
G
21
N
N ALSO HAVE ACCESS TO NORMAL WATERERS
G MUST DRINK FROM GUTTER

Figure 25. Sequence of pens along the gutter. Pigs' water supply

When the trial started the mean weight of the 60 pigs was 57.1 lb. The trial was continued for 36 days. It was to have continued for a longer period; but an outbreak of Transmissable Gasto-enteritis (TGE) occurred at the station and it was recommended that the trial cease, since TGE, as it spread to Unit K, could confuse the results.

Statistical analysis of the results:

Three parameters were measured in the study: weight gain per pen over the 36 day period, feed consumed per pen over the 36 day period and feed conversion efficiency. Two statistical designs using analysis of variance seemed possible, completely randomized design (CR) or randomized complete block design (RCB). Randomized complete block design seemed more reasonable because of the possible effects of pen position along the gutter length.

---

*There is no Figure 26.

The weight gained parameters were analyzed using RCB and the F statistic was not found significant at 5%. Since the degrees of freedom are very low for a RCB of two treatments and three blocks, a test of the relative efficiency of an RCB analysis vs. a CR analysis was performed. This relative efficiency (RE) is given by Ostle (54) as

$$R\ E = \frac{(b - 1)\ B + b(t - 1)E}{(bt - 1)E} \qquad (7)$$

where

b Number of blocks

t Number of treatments

B Mean square, among blocks

E Experimental error mean square

In the case of the weight gained, RE = 0.71 which indicates blocking does not justify the loss of degrees of freedom entailed. Reworking the ANOV as CR, an $F_{1,4} = 12.4$ was obtained, and this was significant at 5% but not at 1%.

The feed consumed was treated in the same fashion yielding a non-significant F (at 5%) in either RCB or CR designs, although R E was only 0.65. However the feed conversion ANOV as RCB produced an $F_{1,2} = 21.2$ which was significant at 5%, but not at 1%. In this case R E = 1.14 indicating that blocking was beneficial.

Table 14. Weight gain, feed and feed conversion. Pigs' water supply

Parameter	Treatment	Result	Standard error[a]
Mean pig weight	G[b]	78.9	--
Over test period lb	N[c]	85.4	--
Daily weight gain per pig	G	1.22	0.028
lb/day	N	1.57	0.028
Daily feed consumed per pig	G	3.69	0.297
lb/day	N	4.27	0.297
Feed conversion efficiency	G	3.04	0.049
lb/lb	N	2.73	0.049

[a] From randomized complete block ANOV.

[b] Must drink from gutter.

[c] Also have access to normal waterers.

Table 14 presents the mean parameters and their standard errors.

Outbreak of TGE, which ended the drinking water trial, seemed a useful opportunity to check the passage of a known pathogen through the system. Through the excellent cooperation of the National Animal Disease Laboratory, it proved possible to monitor the TGE virus. Table 15 indicates the results obtained.

Table 15. Detection of TGE virus in system

Date Day/Mo/Yr	Sample	Dilution[a]	Temperature °F
16/2/71	Manure	10^{-7}	53
16/2/71	Ditch	10^{-4}	52.5
26/2/71	Manure	10^{-3}	53
26/2/71	Ditch	10^{-3}	50.5
24/3/71	Manure	10^{-4}	--
24/3/71	Ditch[b]	10^{-4}	47
28/4/71	Manure	10^{-4}	
28/4/71	Ditch[b]	Not present	60
28/4/71	Lagoon	Not present	--

[a] Greatest dilution at which TGE detectable.

[b] Ditch sealed off, lagoon being recycled.

Lambert[1] of the NADL has indicated that animals may remain carriers of the virus even after complete recovery; this, therefore, would seem to explain the continued detection through into April. TGE is a very resistant virus and remains virulent when kept cool and away from sunlight. Thus it would appear that the treatment system does little to control the virus until the temperature rises. At the time of the last sampling, the ditch was at 60°F. It would also appear that exposing some portion of the ditch through a window might be considered. However, Lambert's opinion is that rats and birds will spread the disease very quickly once it has entered a swine production unit, thus elimination in the treatment system alone might be pointless.

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[1] G. Lambert, Virological Investigation, National Animal Disease Laboratory, Ames, Iowa. Private communication.

Irrigation Disposal 1968

Tile drainage flow quantities:

Irrigation was begun in June 1968 and continued until the following November. The results discussed in this section were obtained from 18th June to 18th September. The irrigation equipment functioned well during the study. There was no noticeable corrosion of the pipe, fittings or pump. A light salt deposit (later analyzed as $Mg(NH_4)PO_4$) appeared on the sprinklers and the pipes, but it did not hamper operation. No nuisance problems from odor or insect breeding were encountered. Even when the lagoon odor was quite strong, this odor dissipated during sprinkling. Little ponding was caused because of the low application rates used. The cover crop was mowed periodically to encourage new growth, but was not removed. This residue, along with the organic material in the applied effluent, created a thin organic mat on the soil surface and this may have been beneficial in dissipating the energy of the water from the sprinklers. Because of drought conditions during part of the test period the supplemental irrigation caused much higher growth rates on the plots than on adjacent untreated areas.

There was no noticeable change in infiltration rate so the initial application rate selected was satisfactory for the entire operation. In preliminary trials, a higher rate had been tried and found to lead to excessive runoff and ponding. Though less than the recommended permissible rate for this soil and cover condition (30), the applied rate was near the maximum possible since minor ponding was observed at the end of an irrigation. Hence, this experience indicates that application rates relating to clear water irrigation should be reduced to 30 - 50% when the irrigation water contains appreciable organic matter. Maintenance of infiltration rate through the test period was attributed to the periodic loading and recovery practiced.

Table 16. Average water quality concentrations from June 18-September 20, 1968. Samples collected 12 hours after start of irrigation

Depth in.	COD mg/l.	Total-N mg/l.	Cl^- mg/l.	P mg/l.	pH
Surface					
1	512	333	110	72	7.8
2	510	334			
3	499	335			
4	532	340			
3					
1	246	164	120	11	7.3
2	225	152	103	8	7.3
3	246	140	110	14	7.6
4	199	158	115	12	7.5
6					
1	269	124	118	14	7.3
2	209	154	105	12	7.5
3	178	182	117	13	7.3
4	175	196	115	9	7.4
12					
1	142	137	112	9	7.2
2	156	103	108	4	7.3
3	202	129	115	11	7.7
4	126	137	117	7	7.3
30					
1	--	--	--	--	--
2	94	91	95	2	7.0
3	104	145	85	3	7.5
4	126	151	94	4	7.3
48-Tile					
1	17	65	75	0.2	7.3
2	30	52	80	0.7	7.2
3	20	54	76	0.4	7.2
4	32	72	79	0.7	7.2

1. 1.5-in. @ 70% available moisture at 6-12 in. deep.
2. 3.0-in. @ 70% available moisture at 6-12 in. deep.
3. 1.5-in. @ 95% available moisture at 6-12 in. deep.
4. 3.0-in. @ 95% available moisture at 6-12 in. deep.

Effluent renovation:

Effluent renovation for all parameters is indicated in Table 16. Removal of COD was judged excellent, with an average of 97% for all treatments. Treatment 3, comprising frequent small applications, seemed the most efficient for decreasing COD. The relation between depth and remaining COD is shown in Figure 27. Approximately one-half of the COD was removed in the upper 3 in. of soil where biological activity is greatest, and physical filtration removed any particulate matter. The COD continued to decrease as the solution percolated through the soil profile.

Removal of N ranged from 79% to 83%, with an average of 80%. Again there was little difference between treatments although Treatment 3 appears slightly superior. The percentage removal of nitrogen was fairly constant throughout the entire period. The relation between applied N concentration and tile drainage N is shown in Figure 28, possible explanations for the decline in applied N discussed in a later section. The applied N from the anaerobic lagoon was about 90% NH_3-N but that draining from the tile at 48 in. depth was essentially all in the form of oxidized N. The change in total N concentration with depth is shown in Figure 29. Table 17 presents the changes in form which occured during percolation. It is postulated that nitrification and denitrification took place as the liquor moved down through the soil. The organic nitrogen applied was filtered out near the soil surface as particulate matter. The oxidized nitrogen in the applied liquor would percolate freely down into the soil with the liquid but NH_3-N was adsorbed on clay particles near the surface

TABLE 17. Form and concentration of nitrogen in sampled soil solution at various depths for treatment 4 - 3.0 inches at 95% available moisture - summer 1968.

Sampled Depth inches	Total-N mg/l	NH_3-N mg/l	NO_3+NO_2-N mg/l	Organic-N mg/l
0-surface	340	285	38	17
6	198	21	174	3
30	135	12	123	--
48-tile	60	1	59	--

as NH_4^+ ions. If this process of adsorption was the only factor affecting the NH_3-N, then the total N in the soil solution below about 6 in. would have been less than 70 mg/l. Also, soil analyses should have indicated a considerable acummulation of N in the surface inches, but none was found. The slightly acidic soil (pH 6.5) made volatilization loss of NH_3 unlikely. Some N was probably assimilated by the cover crop, but the most likely fate of the NH_3-N was nitrification by nitrosomonas to NO_2^-, followed by oxidation to NO_3^- by nitrobacter. Such oxidation would occur when aerobic conditions were restored during the rest period following an irrigation. Lees and Quastel (41) have observed that nitrification in soil does not occur unless the NH_4^+ ion is first absorbed on soil particles.

Initially the N concentration in the soil percolate samples was less than 70 mg/l, but after one month of operation, the concentration in the 6 in. deep samples was about 250 mg/l. This lag in higher N concentrations in the soil solution was attributed to the time needed to increase the population of nitrifying bacteria sufficiently to oxidize the N supplied. Frederick and Broadbent (23) have indicated that

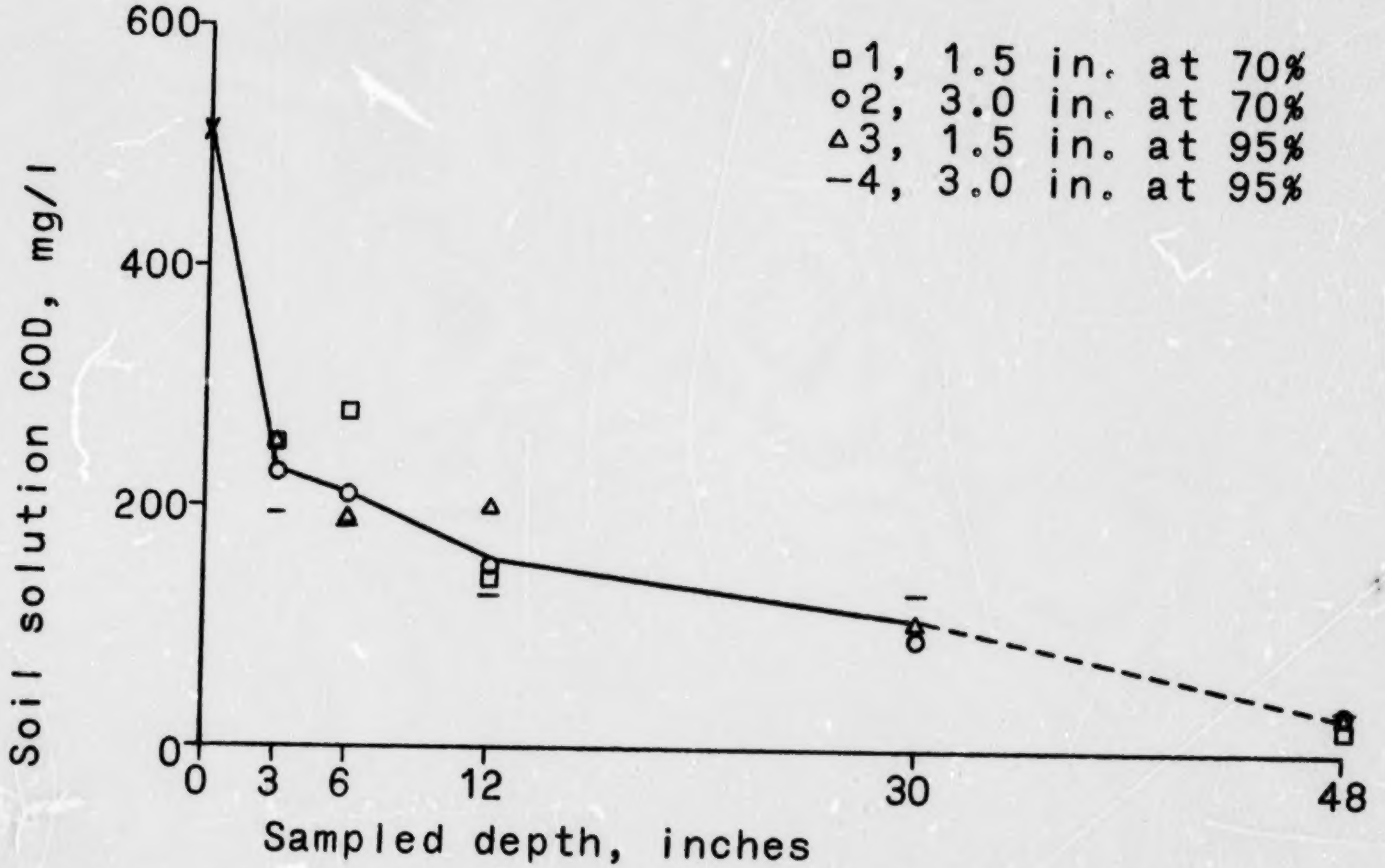

Figure 27. COD concentration in soil solution under application of anaerobic livestock lagoon effluent during summer 1968

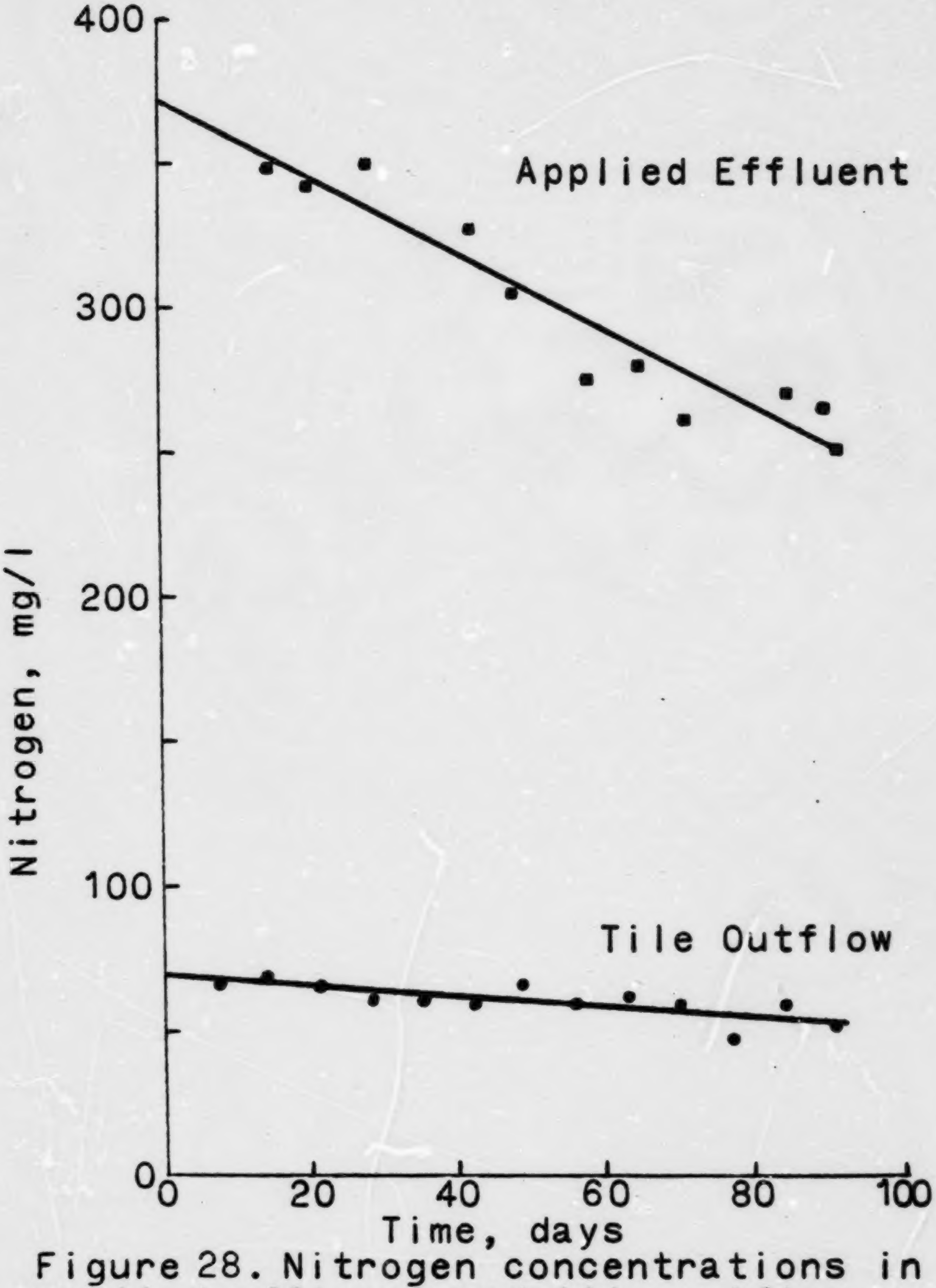

Figure 28. Nitrogen concentrations in applied effluent and tile outflow

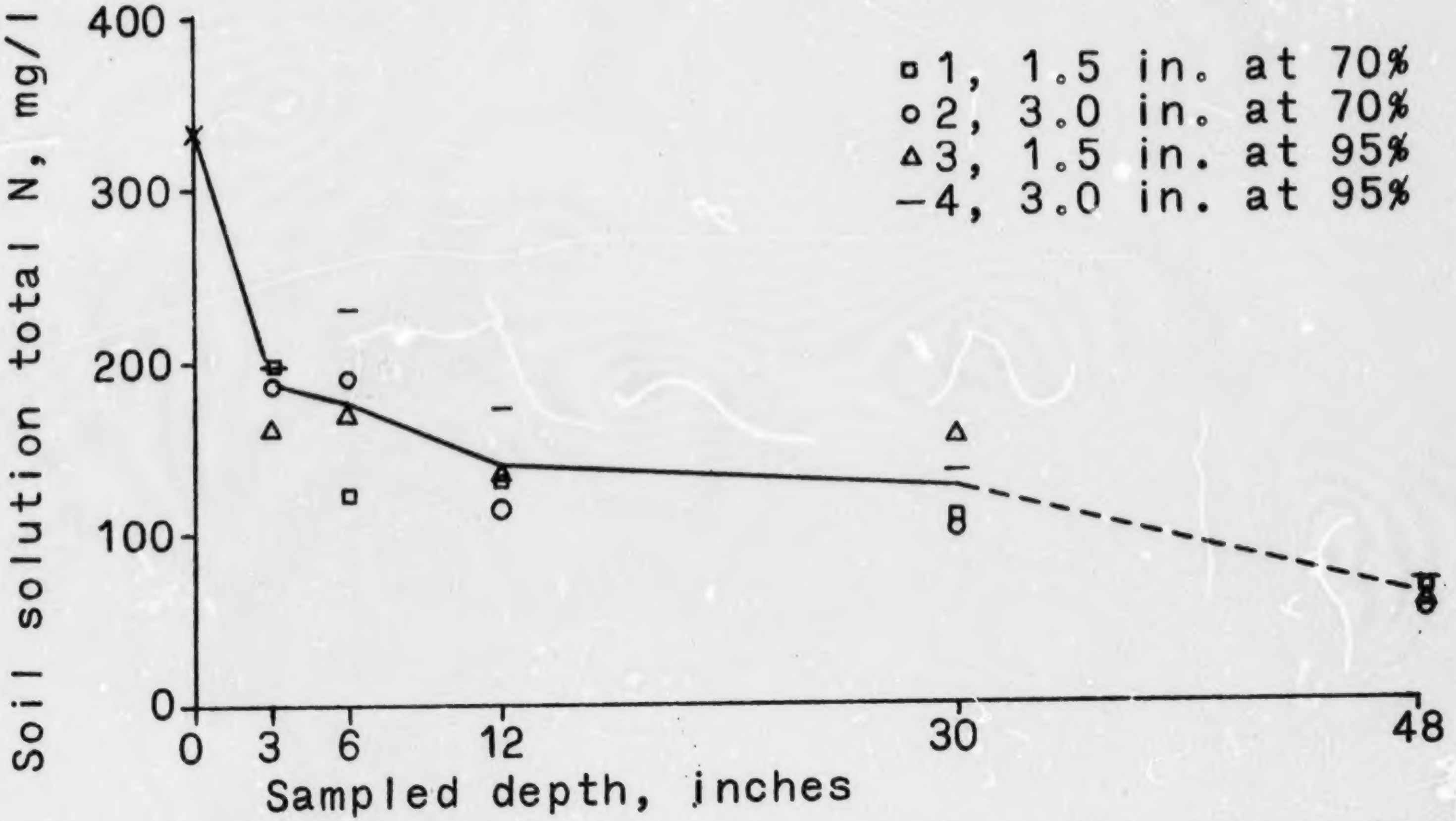

Figure 29. Total nitrogen concentration in soil solution under application of anaerobic livestock lagoon effluent during summer 1968

nitrification becomes very rapid when the population exceeds 10^5 per g of soil and they indicate that lag periods may be expected until such a population is developed.

Much of the N in soil solution samples collected after an application of lagoon effluent originated in earlier applications. This was evidenced by the time lag for nitrification and by the fact that N concentrations in 6 in. samples caused by precipitation were as high, or higher, than those collected after application of lagoon effluent. Samples from the application of effluent after precipitation contained considerably less nitrogen because the percolating rainwater had leached out part of the NO_3^- stored in the upper inches of the soil.

The nitrogen reduction in the soil solution between the 6 in. depth and the tile outlet was about 140 mg/l. The N in this area of the soil profile was almost all in the oxidized form. Uptake by the cover crop at this depth was unlikely, and soil analyses did not indicate an increase in N content of the soil. Therefore this loss must be attributed to denitrification of NO_3^- and NO_2^- to N_2O and N_2 which could escape to the atmosphere (Bremner and Shaw (11)). The nitrifying bacteria in the surface soil produced a good supply of oxidized N during the aerobic rest period after irrigation. This oxidized N was leached downwards by the applied effluent until it encountered anaerobic conditions in the nearly saturated subsoil. Sufficient available carbon, as indicated by COD in Figure 27, was present for denitrification to proceed, until either the substrate or the oxidized N was depleted. A descending aerobic front during a rest period would also inhibit denitrification. Most of the water in the effluent applied from one irrigation remained in the soil profile until more water was added on the surface. Under these conditions, the percolation depth was estimated as being 3 ft, this implied that the period for which the oxidized N and the available C were together in the denitrifying region was at least 5 days. Bremner and Shaw (12) have shown that, if the substrate is readily available, 5 days are quite adequate for extensive denitrification. However, they do show that a poorly available substrate may require up to 20 days for extensive denitrification, thus explaining, in part perhaps, why appreciable COD and oxidized N were always present in the drainage.

The only other likely factor in N reduction in the soil profile would be dilution by foreign drainage from adjacent areas. This was a possibility since the drainage area for each tile was 9000 ft^2 and the plot area was only 2400 ft^2. Table 18 presents an estimate of flow quantities and their relation to the nitrogen balance showing that 86% of the N applied could not be accounted for and was presumed lost by denitrification. The cover crop yield was not determined, but a nitrogen content of 2.5% was measured in three random samples.

All treatments proved very effective in removal of phosphates, each removing 99% or more of the application. The reduction in P concentration was 83% in the top 3 in. of soil, Figure 30. The decrease is attributed to soil absorption. Removal continued as the solution percolated through the profile but the rate decreased with increasing depth.

Table 18. Estimation of nitrogen and chloride reduction by soil filtration considering effects of excess tile drainage 1968-70.

Water, nitrogen, and chloride quantities	1968	1969	1970
Lagoon liquid, inches	30.5	48.0	31.5
Runoff, inches	3.0	2.5	0.5
Precipitation, inches	12.5	31.0	32.0
Runoff, inches	2.0	5.5	3.0
Total water into plots, inches	38.0	71.0	60.0
Estimated evapotranspiration, inches	18.5	33.0	27.0
Expected drainage, inches	19.5	38.0	33.0
Measured drainage, inches	17.5	56.0	46.0
Excess drainage, inches	-2.0	+18.0	+13.0
Nitrogen in total drainage, lb/acre	285	1235	1540
Chloride in total drainage, lb/acre	310	1145	1360
Nitrogen in excess drainage, lb/acre[a]	---	185	135
Chloride in excess drainage, lb/acre[a]	---	220	160
Nitrogen in expected drainage, lb/acre	285	1050	1405
Chloride in expected drainage, lb/acre	310	925	1200
Nitrogen into plots, lb/acre	2070	3140	2740
Chloride into plots, lb/acre	680	1300	1110
Nitrogen reduction, %	86	63	49
Chloride reduction, %	54	29	-8

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[a] Concentrations in excess drainage assumed to be the same as in 6 and 8-ft wells in areas where no liquid was applied, 46 mg/l nitrogen and 54 mg/l chloride.

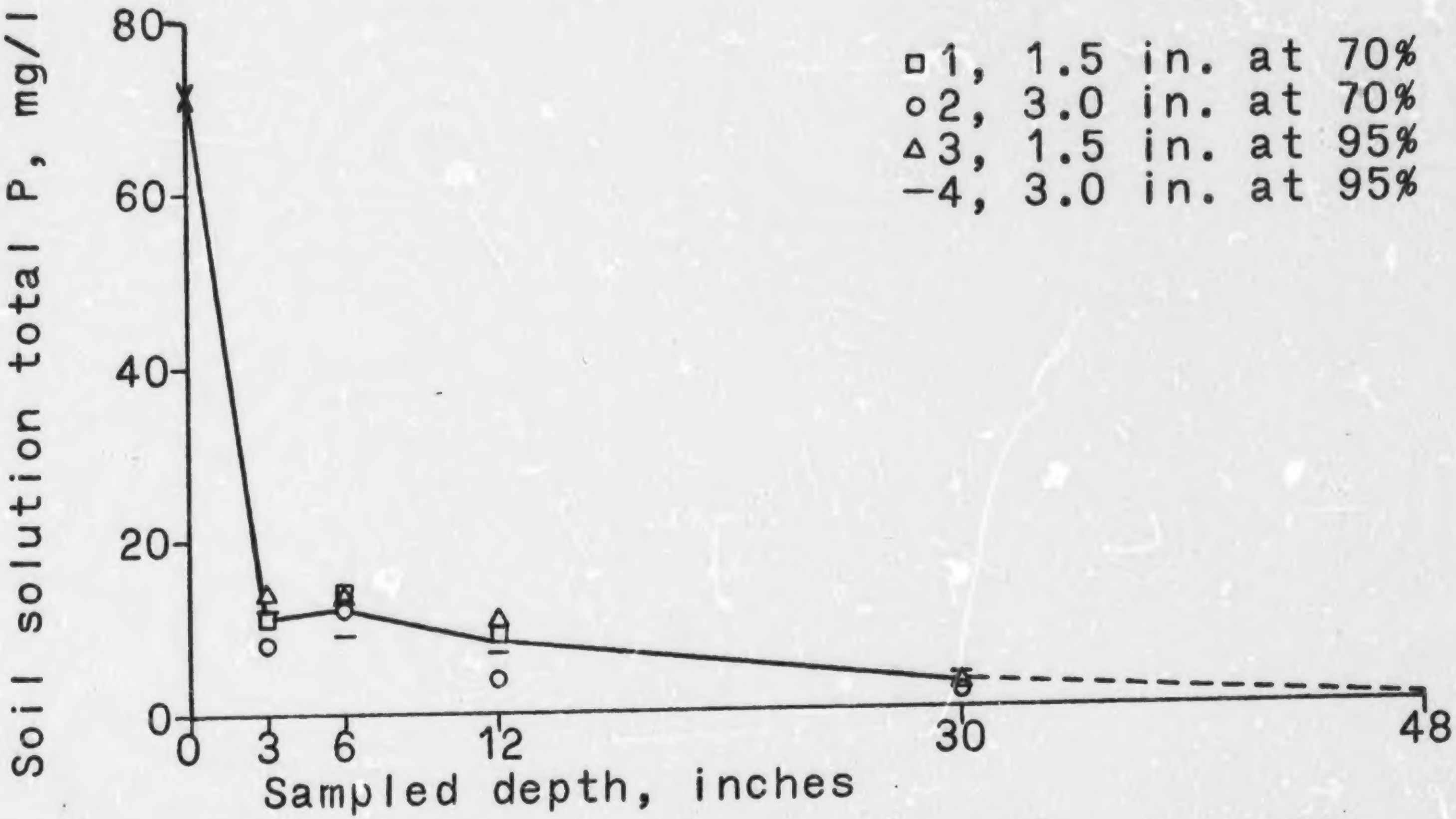


Figure 30. Total phosphorus concentration in soil solution under application of anaerobic livestock lagoon effluent during summer 1968

As can be seen from Table 18, the chloride concentration decreased 54% in the first year. Such a large removal was not entirely expected since Cl is biologically inert, however, the next two seasons' results indicate that the first year's results represent equilibration with the existing soil solution.

Irrigation Disposal 1969/70

Plot condition:

The soil in plots has continued to accept the lagoon liquid as well as or better than during the first year. Equipment traffic on the plots has been kept to a minimum. Some liquid movement on the plot surface occurs because of distribution irregularities, but liquid usually did not move outside the plot boundaries. The irrigation intensity was lower near the edges, and the liquid usually infiltrated. If ponding occured in the low areas, infiltration was reduced or lost until those areas dried.

One major problem in the plots was that the backfilling in the tile trenches settled and cracks developed between the trench walls and the fill material as the soil dried between applications. This allowed short-circuiting of any liquid irrigated over these cracks. Since similar cracks also developed outside the plots, it was concluded that the phenomenon was not related specifically to lagoon water. Yamazaki *et al*. (87) have reported cracks in top soil and surface soil would benefit drainage, but they were not using the soil as a treatment system. No major tillage had been done since the tile installation, and no repairs were attempted until all data collection was complete. Short circuiting caused a marked increase in COD, NH_3-N and total P in the tile drainage. The effective application rate in the immediate area of the tile trenches was about 0.5 iph, which is higher than the recommended rate for application of irrigation water on this type of soil, thus some runoff was expected.

Table 18 shows the treatment schedule for 1970/71. The difference in flow pattern from the tile drains of a typical plot may be seen in Figures 1 and 32; the results of 3 x 2/3 in., application are seen in Figure 31 and they show that the tile drainage concentrations of COD, NH_3-N and total P reach increasingly higher peaks after each application. Figure 32 shows the effect of a single 2 in. application and it is quite apparent that the maximum transient concentration is much higher. Short circuiting resulted as soon as the soil near the tile lines became saturated and drainage started. The quality of this drainage was poor and remained so until the surface was no longer oversaturated. On a concentration basis at peak flow, only about 60% of the COD, 80% of the NH_3-N and 75% of the total P were removed from the liquid applied. On a mass basis, renovation seems much better because only about 20% of the liquid volume applied drained during the first 10 hr after application began. The "regular but often system" did not improve overall COD removal, but it did reduce the peak concentration in the drainage. Total P and NH_3-N removals were better. Seemingly, these nutrients are removed more rapidly than is COD when the liquid is in prolonged contact with the soil.

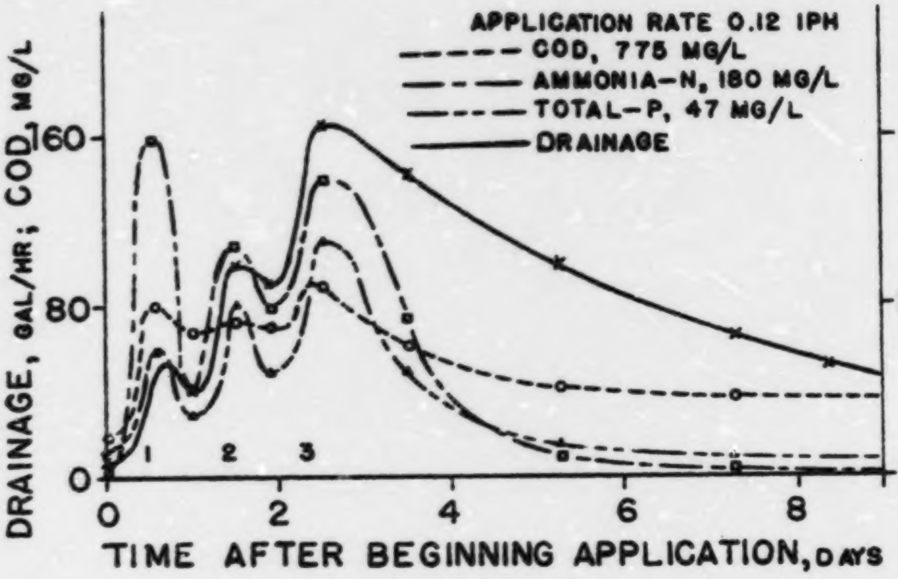


Fig. 31. Tile drainage water quality with 2.0 inches of anaerobic lagoon effluent; applied in 3, 2/3-inch applications.

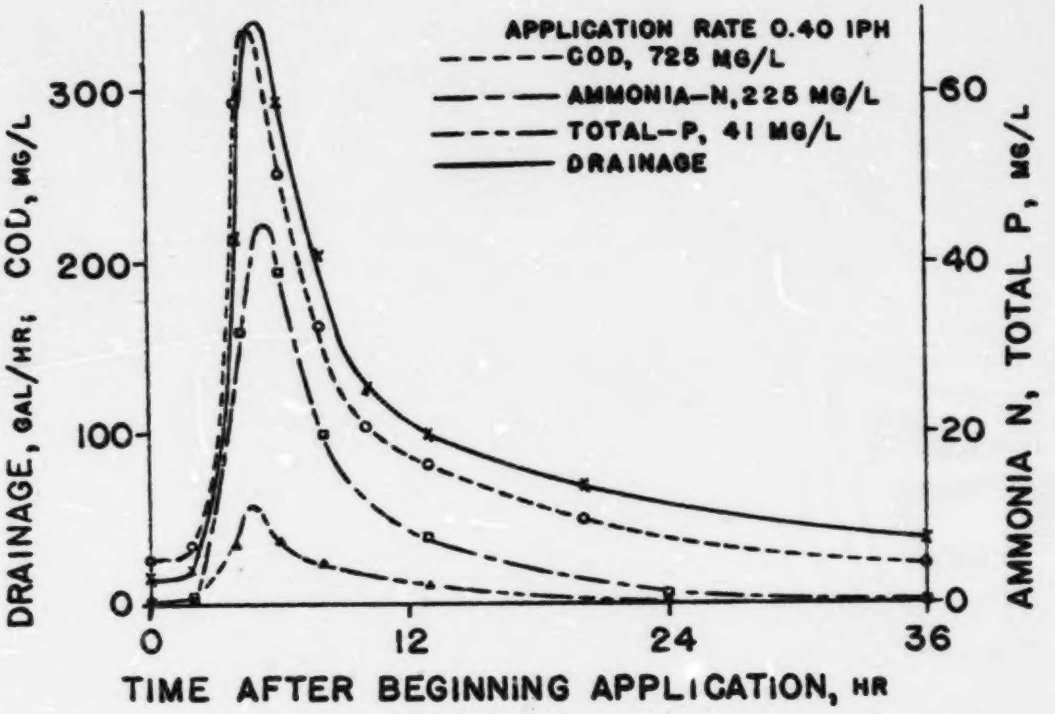


Fig. 32. Tile drainage water quality with one application of 2.0 inches anaerobic lagocn effluent.

Effluent renovation:

In the 1970 field experiments, the mass removal of COD ranged from 79 - 93%. Better removal was obtained with 3 daily applications of 2/3 in. than with a single 2 in. application. As more liquid was applied during the season, removal of COD declined. Samples from the porous cups, which were filtered by both the ceramic material and the soil, indicated reductions in COD of 89% at 6 in., 95% removal at 12 in. and 97% removal at 24 in. As in 1968, removal was most marked in the top few inches of soil where biological activity is greatest. Initial results from laboratory soil columns containing: sand, sand and soil in various proportions and sand, soil and straw showed that 4 ft of coarse sand could remove 90% of the applied COD. Finer textured soils did not remove substantially more COD when the application rate exceed 2 in/wk. Extrapolation of these results to a real soil situation suggests that achieving an effluent strength of 30 - 60 mg/l COD should be feasible.

Almost all the N reaching the tile drainage was in the form of oxidized N. Exceptions caused by short circuiting, which passed appreciable NH_3-N, have been mentioned earlier. The concentration of oxidized N did not change markedly over the season; the same was true for Cl concentration, which was measured to follow water movement and as an indicator of salt buildup in the treatment system.

The concentration of total N in the tile drainage water, Table 18, from the various treatments does not show the whole picture of the N removal by the soil system. Part of the decrease is attributed to dilution by foreign drainage from the buffer areas around the plots.

The excess water in the tile drainage was assumed to have the same concentration of total N and Cl (46 and 54 mg/l, respectively) as the 6 & 8 ft wells in the buffer areas. As may be seen from the Cl removal value for 1970 in Table 18, on a mass basis a point has been reached where the biologically inert soluble salts, such as Cl, are no longer being removed. A condition of dynamic equilibrium has been achieved. At the same time, the total N continues to be decreased by nearly 50%.

Figure 33 shows the N concentration in the 1970 soil system for Treatment 4 (2 in., one time at 350 mb). As in 1968, the N concentration of the lagoon declined as the irrigation system progressed. Also, it should be noted that the surface catch on the plot surface always contained a lower concentration of total N because volatilization was enhanced by the sprinkling process. This finding is somewhat at variance with that of Koch (38) who could not detect N loss from sprinkler water using $(NH_4)HPO_4$ or $NH_4NO_3 + (NH_2)_2CO$ solutions in the pH range 5.5 to 8.5, however, he did find considerable losses using a solution of anhydrous NH_3 in water. Further loss of N taking place in the soil system may have been caused by continued desorption of NH_3-N on the soil surface before infiltration. The amount of total N in the surface catch was used as a basis for calculating N removal by the soil system. Finally, an unknown amount of N was assimilated by the crop.

The quantity of water that moved through the soil profile near the porous cups is not known. When the peak in the concentration of total N at the 6 in. depth occured, little net downward movement of water was occuring because evapotranspiration nearly equalled water addition. After late July, the rest of the season was abnormally wet, and the N moved downward with the excess water. The total N in the soil solution decreased with depth, partly because of dilution. The quantity of water that moved through the soil near the cups could not have been more than the quantity of expected tile drainage from the plots in this treatment (Table 18). Thus, the reduction in concentration was mainly a result of cenitrification.

Laboratory studies have shown that it makes little difference in denitrification rates whether the soil is saturated with lagoon liquid, or with water from precipitation, when the initial NO_3^--N content is 400 mg/l. About 10% of the NO_3^--N added would be removed in 7 days. If no soil was added to the lagoon liquid plus NO_3^--N solution, almost no reduction took place, but if glucose were added at 400 mg/l as an energy source then denitrification began in 2 days. It emerged that 2000 mg of glucose was required to reduce completely 400 mg of NO_3^- in 7 days.

The amount of total P removed is a function of both the application rate and the time in the soil. When the liquid flows through cracks, total P removal is low. Each particular soil has a definite capability for P removal, depending on its type, clay content and organic matter content. In the field experiment, removal ranged from 90 to 97%; cracks were responsible for the poor rates of removal that resulted during the second and third seasons. In the laboratory, sand columns, 3 ft, would cause a loss of 95% of the total P for two months (equivalent to 900 lb/ac). But as the saturation front progressed, the concentration in the leachate

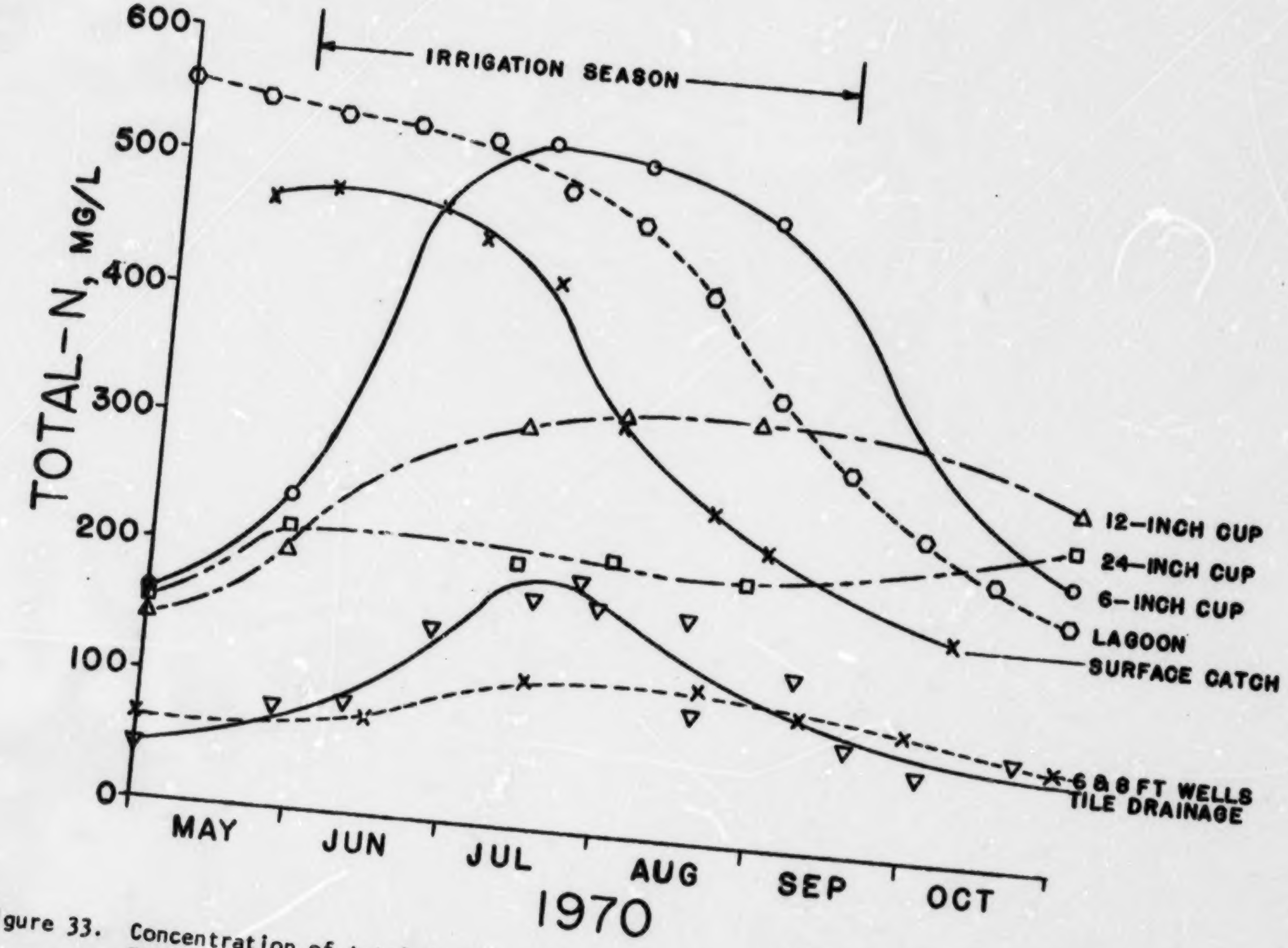


Figure 33. Concentration of total-N at various depths in the soil solution of Treatment 1970-4 on which anaerobic lagoon effluent was applied.

increased until it was half that being applied at 1600 lb/ac equivalent. Greater P reduction in the field would be expected because natural soils have a much greater capacity for P adsorption than does sand.

The total P in the tile drainage can be expected to increase slowly as more liquid is applied. The regions of greater permeability will eventually become saturated, although the bulk of the field may be well below saturation. Figure 34 shows the concentration of total P in the soil solution at various depths in treatment 1970 - 4. Over three seasons, 1700 lb/ac of P had been applied. The concentration at the 6 and 12 in. depths had more than doubled since the first season. The system would still be capable of removing nearly all the total P from the applied effluent if it were not for the cracks. Figure 35 shows the concentration of total P in the tile drainage water caused by significant precipitation after various amounts of lagoon liquid had been applied. The drainage from precipitation did not produce a peak of total P as did the applications; thus, this drainage should be representative of the level of total P that could be expected in the tile drainage water if cracks were absent. There seems a definite correlation between the amount of total P applied and the concentration in the drainage water and, for the moment, a linear relation seems most valid. Linear regression analysis (Ostle (54)) gives

$$\text{Expected total P, mg/l} = (k) \quad \text{total P applied, lb/ac}$$

This relation is tentative and may be lacking other variables such as yearly application rates and annual precipitation.

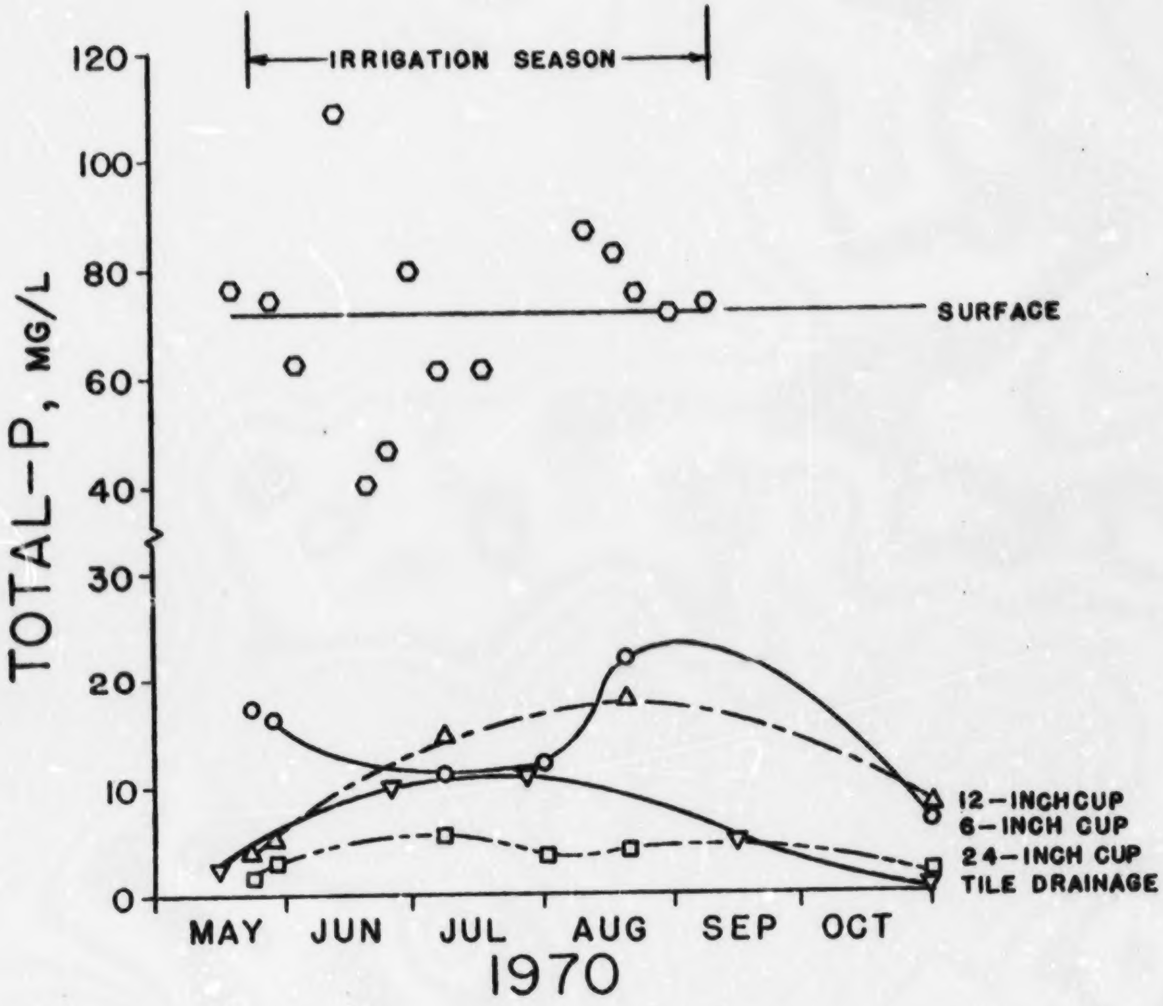


Figure 34. Concentration of total-P at various depths in the soil solution of Treatment 1970-4 on which anaerobic lagoon effluent was applied.

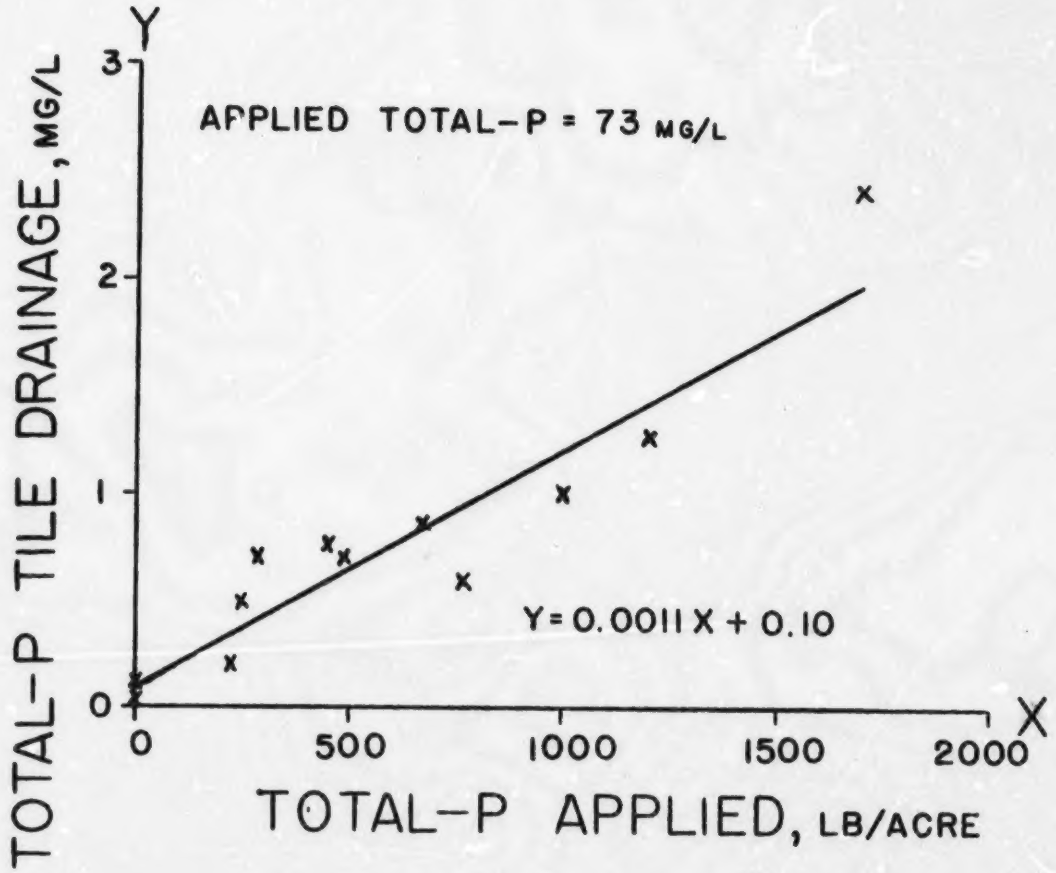


Figure 35. Concentration of total-P in tile drainage water caused by precipitation after various amounts of total-P had been applied in anaerobic lagoon effluent.

Desorption of Ammonia From the Anaerobic Lagoon

Annual variation of nitrogen concentration:

The rate of N influent to the lagoon is approximately constant since the pig population is 500 ± 100 throughout the year. However, earlier sections in this report have alluded to N losses from the lagoon other than those accounted for by liquid abstraction. In the cold weather when bilogical activity is low and an ice cover is present, N removal can only be via precipitation or sedimentation, but during hot weather, loss of ammonia by volatilization is a distinct possibility. This section will discuss some of the theories of NH_3 adsorption and desorption and how these have been correlated with the behavior of the lagoon. Figure 36 shows a typical year's variation in lagoon N concentration.

Nitrogen balance to estimate nitrogen lost by ammonia desorption:

No quantitative method was available to measure the amount of nitrogen lost by desorption. Thus, a nitrogen balance of known quantities of nitrogen added, removed, and accumulated in the lagoon had to be used, and the unaccountable loss was assumed to be by ammonia desorption.

Table 19 shows this balance for the year. In late August, research with the oxidation ditch began, and for the rest of the year, the lagoon received only the overflow from the ditch. The only discharge from the lagoon was either to a temporary holding pond or by irrigation on nearby grassland, so the amount of nitrogen discharged in the excess liquid was closely measured. Seepage losses are very small from this lagoon.

The accumulation of nitrogen within the lagoon is only an estimate. A small fraction of the manure solids are relatively inert and contain some nitrogen. This particular 7-year-old lagoon has shown little solids

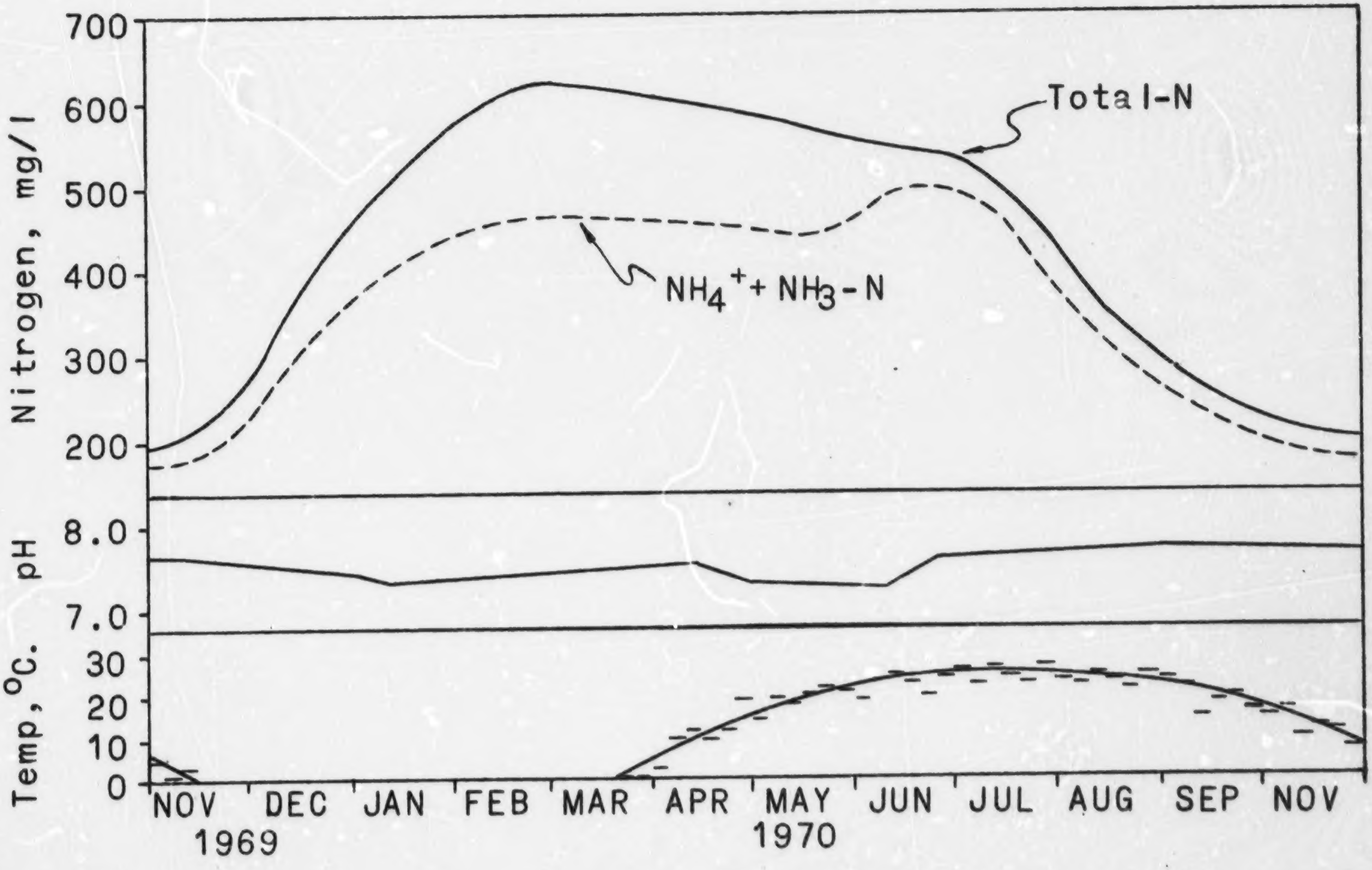


Figure 36. Anaerobic manure lagoon nitrogen concentration, pH, and mean air temperature during 1969-1970 at Ames, Iowa.

Table 19. Nitrogen balance on anaerobic manure lagoon at Swine Nutrition Research Farm, Iowa State University, Ames, Iowa from 1 Nov. 1969 to 31 Oct. 1970.

IN	
Feed: 1 Nov. 1969 to 20 Aug. 1970	
650 hogs at 130 lb avg weight	
85,000 lb total weight at 0.048 lb N/day 100-lb hog	
40 lb/day x 293 days	11,700 lb
21 Aug. to 31 Oct. 1970	
Discharge from oxidation ditch	
2,500 gal at 500 mg/l N	
10 lb/day x 72 days	700 lb
Total IN,	12,400 lb
OUT	
1 Feb. 1970	
Discharge to temporary holding pond	
200,000 gal at 600 mg/l N	1,000 lb
27 Apr. to 1 Nov. 1970	
Research irrigation	
540,000 gal at 485 mg/l N	2,000 lb
15 May 1970	
Field irrigation disposal	
200,000 gal at 500 mg/l N	800 lb
1 Oct. to 31 Oct. 1970	
Field irrigation disposal	
500,000 gal at 220 mg/l	900 lb
Total OUT,	4,700 lb
ACCUMULATION	
1 Nov. 1969 to 31 Oct. 1970	
$MgNH_4PO_4 \cdot 6H_2O$ precipitation	
0.27 lb/day N	+ 100 lb
1 Nov. 1969 to 31 Oct. 1970	
Net volume change	
-200,000 gal at 220 mg/l N	- 300 lb
Organic sludge accumulation	0 lb
Total ACCUMULATION,	- 200 lb

UNACCOUNTABLE = IN - OUT - ACCUMULATION
= 12,400 - 4,700 - (-200)

UNACCOUNTABLE = 7,900 lb N

Note: lb x 0.454 = kg; gal x 3.8 = l.

increase over the past 4 years, and no significant accumulation in solids was likely. A crystalline precipitate of magnesium ammonium phosphate ($MgNH_4PO_4 \cdot 6H_2O$) has been found in this lagoon. The amount of nitrogen in this form depends on the amount of magnesium added. In this lagoon, the loss of nitrogen is believed to be less than 1 percent of the amount added. Other precipitation may be occurring along with other processes that result in nitrogen accumulation, but the amount removed is probably small.

The 7,900 lb (3,600 kg) of unaccountable nitrogen lost is attributed to ammonia desorption and amounts to 64 percent of the added nitrogen.

Ammonia desorption - equilibrium conditions:

In aqueous solution, ammonium dissociates according to equation 8,

$$NH_4^+ \xrightleftharpoons{H_2O} NH_3 + H^+ \qquad (8)$$

Water enters the reaction, also. At equilibrium,

$$\frac{[NH_4^+]\,[OH^-]}{[NH_3]\,[H_2O]} = K_b \qquad (9)$$

and

$$\frac{[H^+]\,[OH^-]}{[H_2O]} = K_w \qquad (10)$$

which yield

$$[NH_3\text{-}N] = \frac{K_w}{K_b}\,\frac{[NH_4^+\text{-}N]}{[H^+]}. \qquad (11)$$

Equation 11 is more easily solved by expressing K_b, K_w, and H^+ as pK_b, pK_w, and pH, respectively. Equation 11 then becomes

$$\left[NH_3\text{-}N\right] = 10^{(pK_b + pH - pK_w)} \left[NH_4^+\text{-}N\right] \qquad (12)$$

The exact concentration of ammonia-N in solution cannot be measured directly; however, Equation 12 makes prediction routine if the ammonium + ammonia-N concentration, liquid temperature, and pH are known. Equation 12 is solved simultaneously with Equation 13 and the percent ammonia-N

$$\left[NH_3\text{-}N\right] + \left[NH_4^+\text{-}N\right] = \left[NH_4^+ + NH_3\text{-}N\right] \qquad (13)$$

in solution at the specified conditions is obtained.

During the year, the temperature of a lagoon surface in Iowa ranges from 0 to 35°C; over this range, pK_b and pK_w change considerable (actual values are readily available in chemistry handbooks). Values of pH from 6.5 to 8.5 might also be expected. Figure 37 shows the percentage ammonia-N in solution over these ranges of temperature and pH. The percentage NH_3-N approximately doubles with either a 10°C rise in temperature or 0.3 rise in pH.

Desorption of ammonia-N:

Volatile components, such as ammonia in aqueous solution, exhibit a vapor pressure because the components seek to establish equal pressure between the two phases (gas-liquid system). With a lagoon open to the atmosphere, equilibrium usually is never reached. The pressure difference between the ammonia-N in the liquid and the air represents the potential or driving force for desorption. This potential is analagous to a temperature difference that causes heat flow.

To evaluate the pressure difference, the partial pressure of ammonia-N in solution must be accurately known. A derivation by Kowalke _et al_. (40),

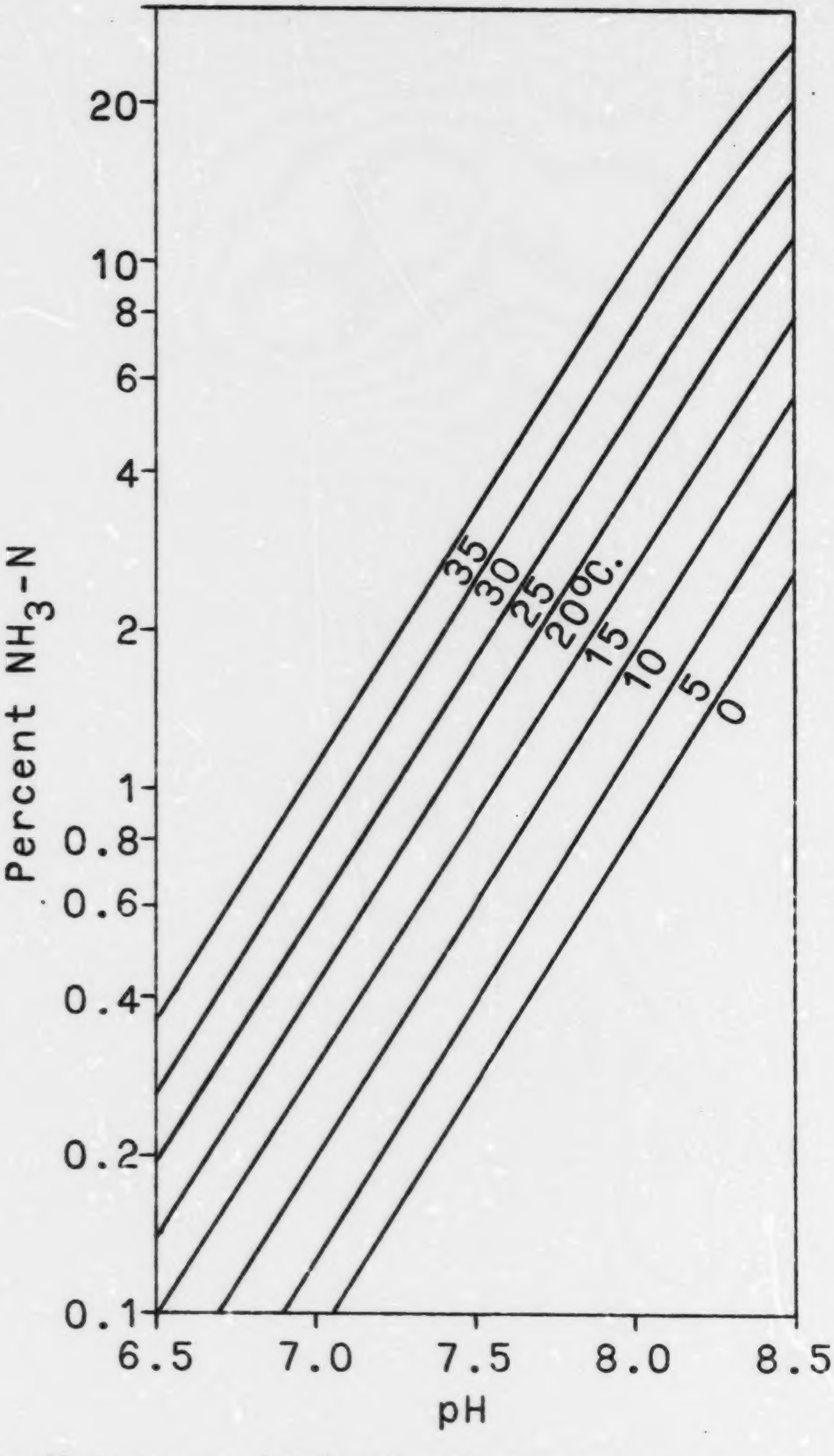


Figure 37. Percentage of NH_4^+ + NH_3-N in solution that is NH_3-N at equilibrium at various expected conditions in an anaerobic manure lagoon.

at standard conditions and 25°C, yields a working relationship,

$$\ln \frac{P_L}{m} = \frac{-4425}{T} + 10.82 \qquad (14)$$

where

P_L = partial pressure of ammonia-N in liquid, in atm

m = molality of ammonia-N in liquid, in moles/l

T = temperature, in °K

This derivation assumes that Henry's law holds over the range in which the equation is used and that the heat of solution of ammonia is independent of temperature, an assumption justified over only a small temperature range.

P_L (ammonia-N pressure in the liquid) is readily calculated from Equation 14 by substituting

$$m = \frac{(NH_3\text{-N\%})(NH_4^+ + NH_3\text{-N, mg/l})}{100\%(14000\ \frac{mg}{mole})} \qquad (15)$$

and solving for P_L. Figure 38 shows P_L during the season as calculated from Equation 14.

The partial pressure, P_g, in the air above the lagoon can be found by measuring the ammonia-N in air samples around the lagoon. The air around the lagoon in this study was sampled frequently during 1970. A measured volume of air is bubbled through distilled water. The absorbed ammonia-N is measured by Nesslerization. P_g is then calculated as

$$P_g = \frac{(NH_3\text{-N})(29)(SVA)}{(AV)(14)(454000)} \qquad (16)$$

where

P_g = partial pressure of ammonia-N in air, in atm

NH_3-N = weight of absorbed ammonia-N, in mg

SVA = specific volume of air at times of test, in cu ft/lb

AV = volume of air sampled, in cu ft

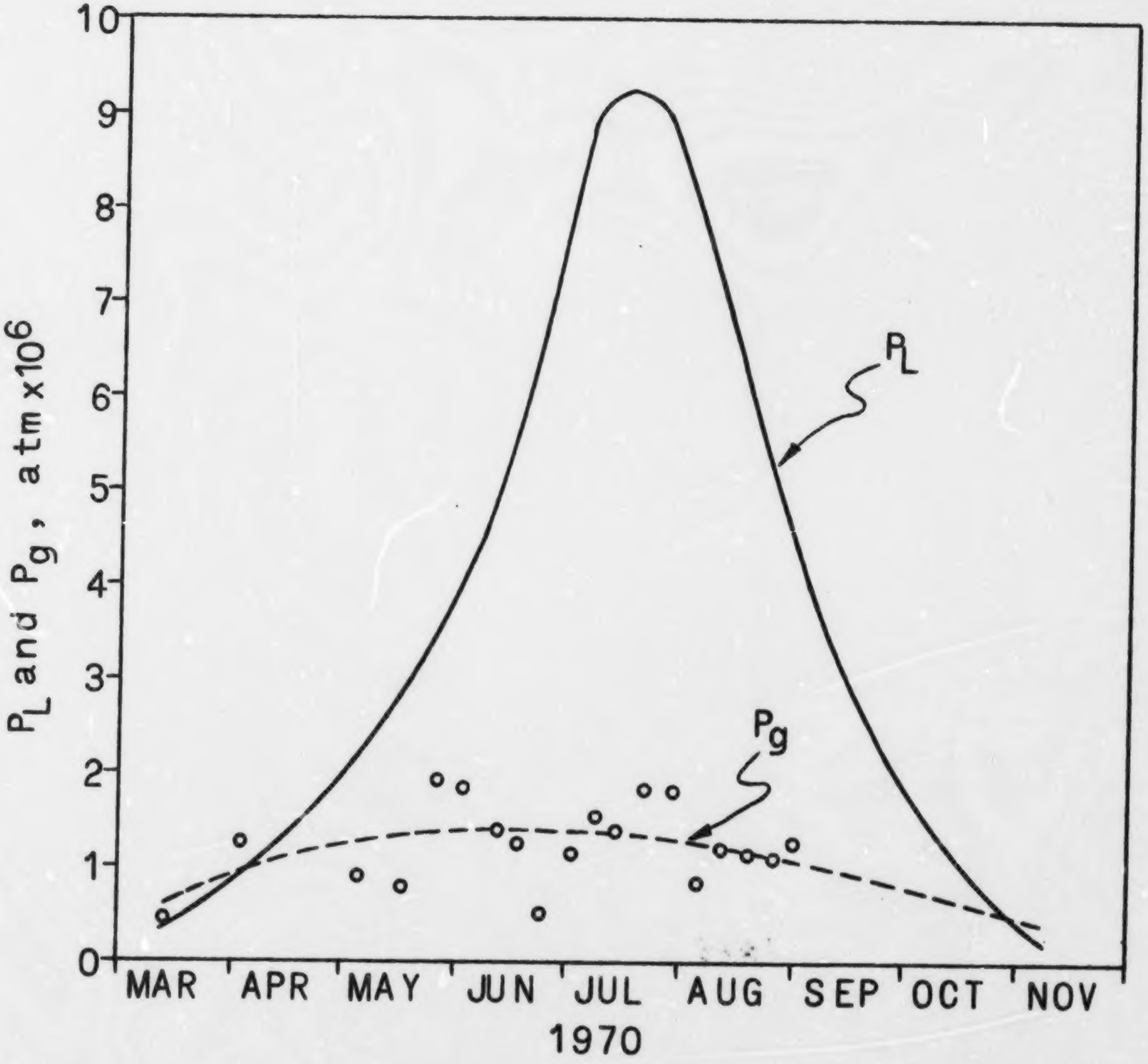


Figure 38. The calculated vapor pressure, P_L, of ammonia-N in an anaerobic manure lagoon and the measured vapor pressure, P_g, of ammonia-N in air directly around the lagoon during 1970 (Note: 1×10^{-6} atm = 0.55 μg/l NH_3-N).

29 = weight of air per mole, in g/mole

14 = weight of ammonia-N gas per mole, in g/mole

454,000 = mg per pound, in mg/lb

The lower curve on Figure 38 is the line of best fit through the points plotted on the graph. These points are the average P_g of the air sampled directly around the lagoon. The sampling locations are 11 - 15 on Figure 39. There is considerable scatter in the data; the different climatic conditions may well be responsible. No consistent pattern was established. P_g was usually higher when the wind velocity was very low. At sampling points 9 and 10, the P_g's usually were less than half the average P_g very near the lagoon.

Mass-transfer Considerations:

The familiar mass-transfer relation is assumed to apply for the desorption process,

$$\frac{d(NH_3\text{-}N)}{dt} = AK(P_L - P_g) \qquad (17)$$

where

NH_3-N = weight of ammonia-N transferred, in lb

t = time, in days

A = area of liquid-gas interface, in sq ft

K = over-all transfer coefficient, in lb/day-sq ft-atm

P_L = partial pressure of ammonia-N in liquid, in atm

P_g = partial pressure of ammonia-N in air, in atm.

All the parameters in Equation 16 except K, the over-all transfer coefficient, can be evaluated from collected data and previous equations.

The mass transfer in this case is analogous to heat transfer from liquid to gases. There are two films that offer resistance to transfer of ammonia-N just as there are two films that offer resistance to heat flow.

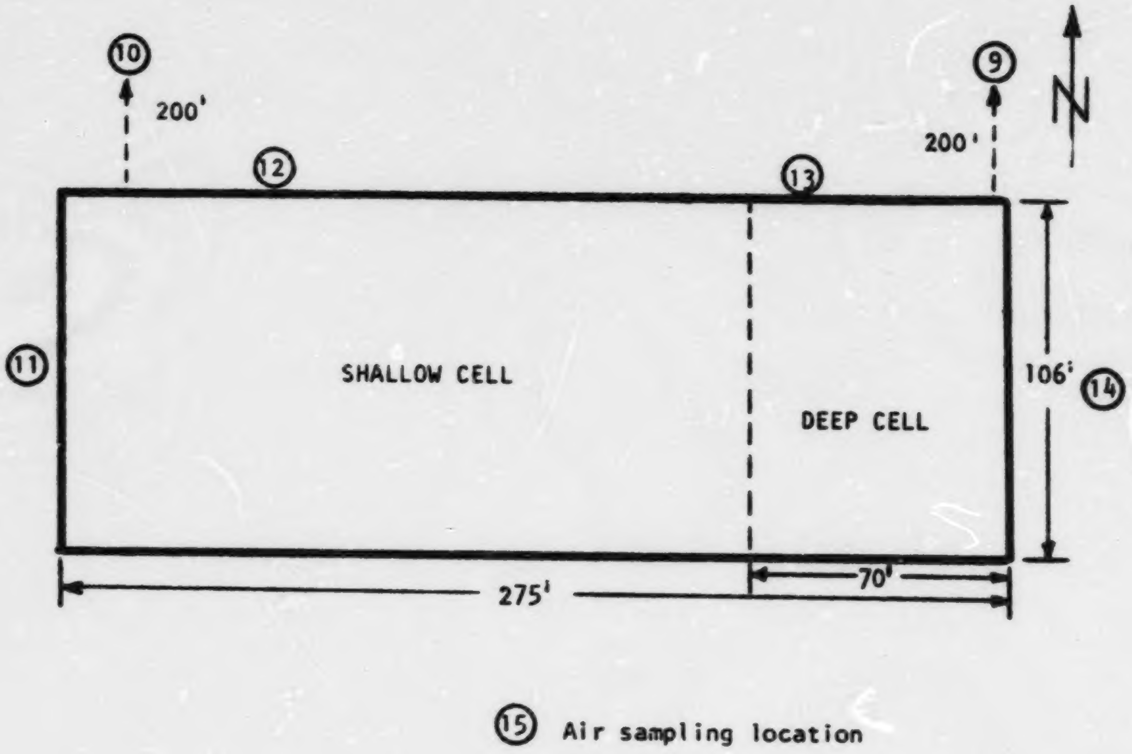

⑩
200'
⑨
200'
N
⑫
⑬
⑪
SHALLOW CELL
DEEP CELL
106'
⑭
70'
275'
⑮ Air sampling location

Figure 39. Air sampling location around the anaerobic manure lagoon at Swine Nutrition Research Farm

In absorption, the overall transfer coefficient is the sum of transfer coefficients of the separate films. In series, the resistances are additive and equal to the reciprocal of the overall transfer coefficient:

$$\frac{1}{K} = \frac{1}{K_a} + \frac{1}{K_l} \qquad (18)$$

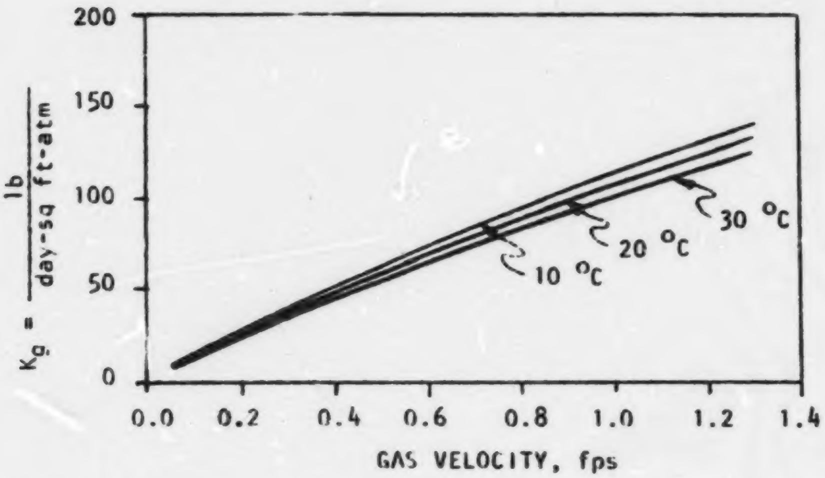
where

K_a = transfer for air film

K_l = transfer for liquid film.

A classical experiment to measure the absorption of ammonia into water was done by Halsam et al. (26). Water flowed down the walls of a column, and an ammonia-air mixture was blown up through the tower. The experiment was done at different gas velocities and temperatures. Figure 40 represents these data.

Figure 40. Ammonia transfer coefficient versus gas velocity for ammonia absorption into water from Halsam et al.



For ammonia, the gas film fully controlled the value of $K_g = K$ (in this paper) for low velocities. Further work by the same authors on the effect of gas velocity and temperature was done with these data. The equation for K was found to be

$$K = 2.6 \times 10^5 \, V^{0.8} \, T^{-1.4} \, \frac{\text{lb}}{\text{day-sq ft-atm}} \qquad (19)$$

where

V = gas velocity, in ft/sec

T = temperature, in °K.

Other work by Whitman and Davis (83) found K = 67 at 20°C for an ammonia-air mixture passing over a free-water surface with the water stirred at a constant rate. Absorption was controlled by resistance of the gas film. Kowalke et al. (40) found K = 1.6 for an ammonia-air mixture being absorbed from quiet air at 25°C.

Equation 19 will hold for desorption if the same conditions that control absorption are met. These conditions are that the liquid film has the same P_L as the bulk liquid and that all the pressure drop $(P_L - P_g)$ occurs across the air film. If only the diffusion of ammonia-N through the bulk liquid is considered, it would be difficult to maintain the liquid film at or near equilibrium. But diffusion is not the only process, the liquid film probably is replenished mainly by ionization of ammonium to ammonia according to Equation 8. The diffusion of ammonium need only be about 1/60 to 1/100 as great as diffusion of ammonia to the liquid film. The pH may be slightly lowered at the film because of ionization. The acceptance of K for absorption being nearly the same for desorption from a lagoon seems plausible.

Evaluation of K:

To determine if K can be predicted from Equation 19 by using surface conditions of mean air temperature and average wind velocity, the data from 16 March to 3 Oct. 1970 were divided into half-month increments, and K was evaluated for each increment. For the same increments, K was also calculated by using mass-transfer data for the lagoon in Equation 17.

For the mass-transfer evaluation, one estimate had to be made of the amount of nitrogen in undigested and settleable solids that were stored during cold conditions and released during the warm season. The amount stored and released was about 600 lb. These estimates are in column B of Table 20. The amount of change in nitrogen in solution in column A takes into account the change in concentration and the change in volume over each increment because of rainfall, irrigation withdrawal, evaporation, and addition from waste flow. The feed rate, column C, was constant up to late August when the oxidation ditch treatment system was put into operation. From then on, the lagoon received only the overflow from the ditch. The nitrogen lost in column D is the unaccounted for difference between inputs and withdrawal. This loss is considered to be the amount that was desorbed over the increment. To complete the list of needed variables in Equation 17, $(P_L - P_g)$ was picked from Figure 38 for each increment. The surface area of the lagoon averaged about 29,000 sq ft.

K from mass-transfer considerations in column E are the final results of the mass-transfer calculations. During periods of low temperature, $(P_L - P_g)$ is very small or negative, so K has no meaning. At low temperatures either other processes are involved, or some of the underlying equations break down.

Table 20. Prediction of the over-all mass-transfer coefficient of ammonia-N, K, from mass-transfer considerations and conditions prevailing at the surface of an anaerobic manure lagoon during 1970.

Time increment[a] 1970	A N added or lost from solution, lb/day	B N stored in or released from settled solids, lb/day	C N in manure fed, lb/day	D N lost by apparent desorption, lb/day	E K from mass-transfer considerations lb/day-sq ft-atm	F Ave. mean air temp, ^oC	G Ave. wind velocity fps	H K from surface conditions, lb/day-sq ft-atm
032	+2	+6	40	32	neg 1840[b]	1	9.5	620
041	+2	+5	40	33	neg 1065[b]	7	9.5	588
042	0	+1	40	39	2635 752[b]	13	8.9	547
051	+15	-4	40	29	933	17	8.8	530
052	+8	-10	40	42	724	20	7.8	473
061	+5	-10	40	45	431	21	7.5	457
062	0	-7	40	47	290	21	7.2	442
071	-10	-4	40	54	252	23	4.8	320
072	-15	-3	40	58	271	24	5.1	333
081	-17	-1	40	58	327	22	3.5	249
082	-14	0	20	34	279	21	3.8	267
091	-6	0	10	16	187	18	6.6	419
092	-8	0	8	16	394	16	4.5	314
101	-4	0	8	12	690 295[b]	13	7.9	498
102	-4	+2	8	10	3450 492[b]	9	5.4	374

[a] first two digits represent the month and the last digit signifies first or second half of the month.

[b] assumes the partial pressure of ammonia-N above the lagoon is 0.

No data were taken on the lagoon for temperature and wind velocity that could be used in Equation 19 for K from prevailing surface conditions. So, data from the recording station Ames 8WSW (78) were used. The lagoon is 6 miles east of the station. The mean air temperature for each increment was assumed to be the temperature at the liquid-air interface in column F. The wind velocity at the station is measured approximately 4 ft above the ground surface so that should be representative of the velocity at the lagoon surface in column G.

K from prevailing surface conditions in column H are the final results of the calculations from Equation 19. This use of Equation 19 may be open to question because the velocities used to determine it were from 0.2 to 1.4 fps, while it has been extrapolated in this case to velocities as high as 9.5 fps. But from period 52 through 92, the two K's compare quite well. The average K from mass-transfer considerations is 355 and average K for prevailing surface conditions is 364. Much of the desorption occurs during this period. So, the amount of desorption from this lagoon can be estimated if the ammonium + ammonia-N, pH, and surface area of the lagoon are known along with the climatological data for the area.

Laboratory study:

Another phase of this work involved desorption under controlled conditions in the laboratory. Several 500-ml samples of lagoon liquid taken in late November were placed in 1-liter flasks. Air was drawn into the flask through a glass tube located from 0.5 to 2.0 in. above the surface of the liquid. The air leaving the flask was bubbled through weak boric acid to catch the ammonia-N. The nitrogen concentration in the lagoon liquid and boric acid solution were measured periodically by Nesslerization of a small sample. The recoveries in the boric acid were from 95 to 105 percent of the measured amounts desorbed from the lagoon liquid. The desorption from four of these runs are shown in Figure 41, along with temperature, pH, and the resultant P_L for Run No. 4.

The desorption process was nearly linear over time. This was caused by a rise in pH over time that cannot be explained. Such a rise likely would not occur in the lagoon. For Run No. 4, the P_g averaged 11.5×10^{-6} atm and P_L averaged 12.2×10^{-6} atm over the run. This resulted in K from mass-transfer considerations of 122. K from prevailing surface was not measurable. These data support the previous findings that nitrogen does escape from the lagoon liquid by ammonia desorption.

Implications of ammonia desorption:

The loss of about two-thirds of the nitrogen to the air by desorption is important in several respects:

First, the anaerobic lagoon is a major source of nitrogen for the nearby environment. Enough escaped during 1 year from this lagoon to adequately fertilize 60 acres of corn with nitrogen. The desorption is similar to overfertilization of nearby lakes with nitrogen from cattle feedlots. The concentration of ammonia in the air is of little consequence unless a lake is nearby.

Second, there is now a major constituent of the gases from this lagoon that can easily be measured. Possible, ammonia concentrations in air can be correlated to odor levels in a quantitative evaluation.

Third, and probably the most important result of this reduction of nitrogen in the lagoon liquid, is the reduction of land needed for irrigation disposal of the excess lagoon liquid. A reasonable criterion for how

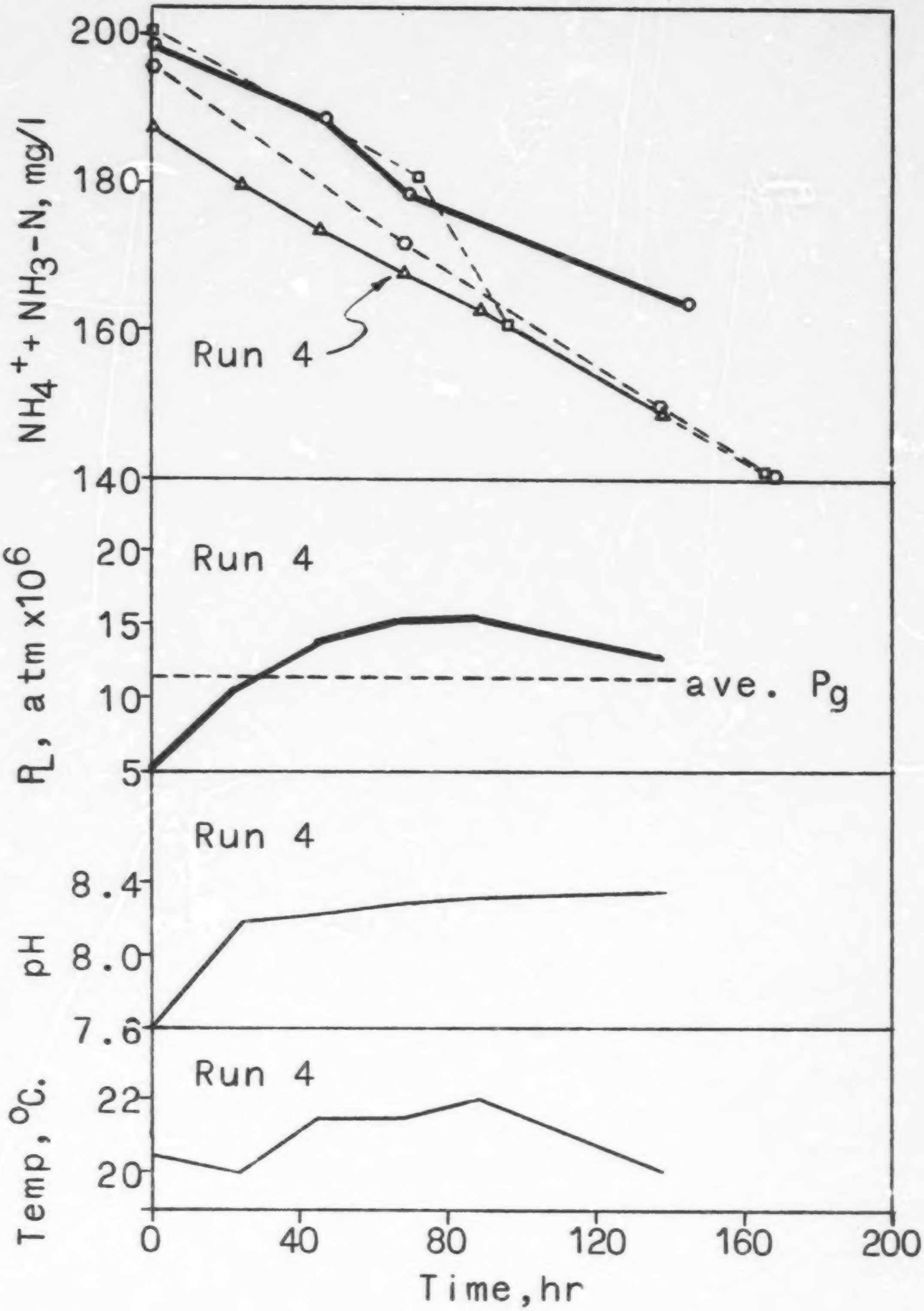


Figure 41. Desorption of ammonia-N from anaerobic manure lagoon liquid in laboratory by drawing air over the liquid surface and catching desorbed ammonia-N in boric acid.

much liquid to apply per year appears to be an amount containing up to 600 lb of nitrogen per acre. The ammonia desorption rate not only dictates how much land will be needed, but also gives the best time of the year for disposal. If disposal can be delayed until Aug. to Oct., only about two-thirds as much land would be needed as compared with uniform disposal from May to Oct. During application of the lagoon liquid with sprinkler irrigation equipment, more nitrogen is lost by desorption, from 15 to 30 percent of the nitrogen pumped from the lagoon lost by the time the liquid reached the soil surface.

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APPENDIX A. CHEMICAL ANALYSES DIFFERING FROM STANDARD METHODS

Biochemical Oxygen Demand

The non-nitrifying BOD test using N-Serve[3] (2 - chloro - 6 - (trichloromethyl) pyridine) has been outlined by Young (88). In this reference 10 mg/l of N-Serve is recommended in the dilution water, however since the chemical is only soluble to 40 mg/l at 68°F, a saturated solution was substituted. Dilutions of this saturated solution were tried but the saturated solution did not appear to inhibit the carbonaceous demand. It is admitted that these tests were not very comprehensive, but Young (88) indicates that N-Serve concentration is not critical because of its low solubility.

Solids

Solids determination by Standard Methods (4) were not satisfactory because the excess of fine material in most samples plugged any filter before a reasonable volume could be passed through. The technique developed was to measure total solids conventionally and also to measure solids on the filtered (8μ membrane) supernatant from a swinging arm centrifuge. The difference between the two was regarded as suspended solids. The method was subject to some errors. The centrifuge tubes had no caps and hence some evaporation may have taken place during centrifugation, also, the centrate was never wholly clear. An angle head centrifuge of greater "g" rating would be an improvement.

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[3] Registered trademark of the Dow Chemical Company.

Kjeldahl Nitrogen

This parameter was determined by the Standard Methods (4) technique until part way through Phase 2. The accepted method works well but seems prone to froth very vigorously when adding concentrated NaOH prior to distillation and occasionally the frothing would be sufficient to contaminate the boric acid trap. Alternatives to the mercuric ion used as a catalyst are selenium and copper. Copper selenite was available in the laboratory and about 0.2 g/10 ml conc. H_2SO_4 was tried in place of mercuric sulphate. Raising the pH after digestion proved a much more docile process and no quantitative differences in nitrogen recovery were obvious. Since concluding the laboratory work, the use of Se and Cu as catalysts has been better documented. Streuli and Averell (74) discuss the use of Hg, Se and Cu and they conclude that Hg is the most effective catalyst since it catalyzed the digestion of heterocyclic compounds but Se and Cu used in the correct proportion are more convenient for many substances. Future laboratory work is needed to determine the efficiency of Se or Cu vs. Hg with respect to the liquors encountered in animal waste treatment.

Nitrate Nitrogen

Nitrate nitrogen was reduced to nitrite by cadmium and measured spectrophotometrically. The method used encapsulated commercial reagents.[4] A standard nitrate solution was used periodically to check the accuracy.

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[4] Hach Chemical Company.

No reference to this cadmium reduction method has been found in the familiar literature.

Chlorides

Chlorides were measured using the mercuric nitrate/diphenylcarbazone method, when this test was first tried in preliminary work very indistinct endpoints were obtained. Knowing that mineral supplements are fed in the form of sulfates it was thought that some might be reduced to sulfite, and Standard Methods (4) lists this as an interference. To overcome such interference the samples were digested on a reflux rack for 30 min. The reagents added to 50 ml of sample were 50 mg $Ba(NO_3)_2$ and 0.2 ml fuming HNO_3. Although this technique improved precision markedly, it has since been found redundant when the sample is first centrifuged to remove organic particles.

Phosphate

The method used was a commercially available modification of the ascorbic acid method of Murphy and Riley (48). The types of phosphate measured were ortho, soluble but able to be hydrolyzed by acid, and total phosphate. The PO_4^{-3} is measured spectrophotometrically and may range from 0 - 3 mg/l. Because the phosphate levels encountered were often greater than 500 mg/l rather extreme dilutions had to be made. The reagent for the acid hydrolyzed phosphate is a measured amount of a powdered acid and the sample is boiled in a water bath for 30 min. Total organic PO_4^{-3} is determined by using a measured amount of acid and persulfate and refluxing the sample for 30 min. After cooling the sample pH is adjusted. The final step in both procedures is the determination of the hydrolyzed PO_4^{-3}. Even when taking great care with the initial sample dilutions, considerable difficulty was encountered with the total phosphate. The determination would quite often be less than the orthophosphate value on the same sample. Standard Methods (4) did not give any clue. Although tannin, lignin and silicates are said to interfere, these were not thought to be present in appreciable quantities after the 1:500 or 1:1000 sample dilution. Finally better total organic results were obtained by taking:

2 gm	$K_2S_2O_8$
5 mg	conc. H_2SO_4
5 ml	raw sample
18 ml	H_2O

This mixture was refluxed for 30 min on a boiling rack. After cooling the mixture would be diluted to 1:500 or 1:1000. This further dilution was extreme enough to obviate the need for neutralization. Finally the phosphate was measured using the spectrophotometric method.

APPENDIX B. RAW RESULTS PHASE 1

Table B1. Nitrogen forms, manure slurry

Date Day/Mo/Yr	Kjeldahl mg/l	Ammonia mg/l	Organic mg/l	Nitrate mg/l
3/2/69	1,480	630	850	0
10/2/69	1,590	505	1,085	0
17/2/69	1,300	420	880	0
24/2/69	--	--	--	--
3/3/69	1,790	840	950	0
10/3/69	1,430	730	700	0
17/3/69	1,410	476	934	0
24/3/69	1,710	545	1,165	0
31/3/69	1,090	350	740	89
7/4/69	855	392	463	78
14/4/69	1,020	280	740	50
21/4/69	1,340	399	941	28
28/4/69	644	133	511	164
5/5/69	830	164	666	178
12/5/69	--	--	--	--
19/5/69	1,190	252	938	220
26/5/69	1,580	448	1,132	228

Table B2. Nitrogen forms, lagoon

Date Day/Mo/Yr	Kjeldahl mg/l	Ammonia mg/l	Organic mg/l	Nitrate mg/l
3/2/69	600	500	100	0
10/2/69	615	268	347	0
17/2/69	626	435	191	0
24/2/69	--	--	--	--
3/3/69	610	560	50	0
10/3/69	625	445	180	0
17/3/69	580	451	129	0
24/3/69	395	305	90	0
31/3/69	465	354	111	0
7/4/69	373	312	61	0
14/4/69	435	333	102	0
21/4/69	430	320	110	4
28/4/69	420	312	108	10
5/5/69	570	240	330	14
12/5/69	--	--	--	--
19/5/69	493	450	43	20
26/5/69	485	462	23	20

Table B3. Nitrogen forms, returned effluent

Date Day/Mo/Yr	Kjeldahl mg/l	Ammonia mg/l	Organic mg/l	Nitrate mg/l
3/2/69	470	425	--	0
10/2/69	493	336	157	0
17/2/69	519	400	119	0
24/2/69	--	--	--	--
3/3/69	455	468	--	0
10/3/69	667	525	142	0
17/3/69	501	451	50	0
24/3/69	428	366	62	0
31/3/69	368	280	150	130
7/4/69	232	235	--	164
14/4/69	201	160	41	206
21/4/69	210	195	15	24
28/4/69	90	43	47	198
5/5/69	85	32	53	190
12/5/69	--	--	--	--
19/5/69	119	56	63	412
26/5/69	288	91	197	356

Table B4. Phosphate (as PO_4^{-3})

Date	Lagoon		Returned poly	Effluent total
Day/Mo/Yr	Poly mg/l	Total mg/l	mg/l	mg/l
3/2/69	192	187	73	100
10/2/69	205	245	84	96
17/2/69	230	310	93	143
24/2/69	--	--	--	--
3/3/69	228	340	93	159
10/3/69	278	410	120	170
17/3/69	286	308	116	180
24/3/69	160	220	110	154
31/3/69	214	278	96	318
7/4/69	200	330	80	150
14/4/69	280	274	128	226
21/4/69	350	384	154	192
28/4/69	324	444	748	380
5/5/69	530	507	660	523
12/5/69	--	--	--	--
19/5/69	428	360	616	820
26/5/69	440	564	468	564

Table B5. pH, temperature and coliforms, manure slurry

Date Day/Mo/Yr	pH	Temp °F	Coliforms[a] N°/100 ml
3/2/69	--	41	--
10/2/69	8.1	35.5	--
17/2/69	--	42.5	3.3×10^7
24/2/69	--	--	--
3/3/69	8.0	42.5	2.1×10^7
10/3/69	--	43.5	0.33×10^7
17/3/69	7.9	48	2.4×10^7
24/3/69	7.7	48.5	2.6×10^7
31/3/69	7.8	43	0.78×10^7
7/4/69	7.5	60	7.0×10^7
14/4/69	7.3	56	4.7×10^7
21/4/69	7.3	61	2.3×10^7
28/4/69	7.0	55	2.1×10^7
5/5/69	6.9	71	--
12/5/69	--	72	5.3×10^7
19/5/69	6.9	64	1.9×10^7
26/5/69	7.2	70	1.9xx 10^7

[a] Membrane filter technique.

Table B6. pH, chloride, temperature and coliforms, lagoon

Date Day/Mo/Yr	pH	Cl^- mg/l	Temp °F	Coliforms[a] N^o/100 ml
3/2/69	--	--	36	--
10/2/69	7.4	--	32	--
17/2/69	--	--	35.5	120 x 10^5
24/2/69	--	--	--	--
3/3/69	7.4	--	35	7 x 10^5
10/3/69	--	--	35.5	6.5 x 10^5
17/3/69	7.5	--	37	8.7 x 10^5
24/3/69	7.5	144	37.5	8.2 x 10^5
31/3/69	7.4	163	35	7.1 x 10^5
7/4/69	7.4	155	54.5	4.2 x 10^5
14/4/69	7.6	168	55	3.9 x 10^5
21/4/69	7.5	174	58	2.6 x 10^5
28/4/69	7.3	168	57	1.8 x 10^5
5/5/69	7.3	144	65.5	--
12/5/69	--	--	69	6.7 x 10^5
19/5/69	7.3	186	61	1.6 x 10^5
26/5/69	7.3	184	69	18 x 10^5

[a] Membrane filter technique.

Table B7. pH, chloride, temperature and coliforms, returned effluent

Date Day/Mo/Yr	pH	Cl^- mg/1	Temp °F	Coliforms[a] N°/100 ml
3/2/69	--	--	34.5	--
10/2/69	8.3	224	32	--
17/2/69	--	234	36	0.85×10^5
24/2/69	--	--	--	--
3/3/69	8.1	226	37.5	0.08×10^5
10/3/69	--	240	35	0.51×10^5
17/3/69	8.2	245	37	0.3×10^5
24/3/69	8.2	236	39.5	0.41×10^5
31/3/69	8.0	188	35.5	3.1×10^5
7/4/69	7.9	177	45.5	0.3×10^5
14/4/69	7.8	173	52	2.0×10^5
21/4/69	7.7	172	56	2.1×10^5
28/4/69	6.3	162	56	0.85×10^5
5/5/69	5.6	164	62	--
12/5/69	--	--	61	--
19/5/69	6.2	179	61	0.055×10^5
26/5/69	6.8	179	66	0.38×10^5

[a] Membrane filter technique.

Table B8. Total solids, manure slurry

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	13,290	9,970	3,320
10/2/69	17,760	13,830	3,930
17/2/69	13,450	--	--
24/2/69	--	--	--
3/3/69	16,390	12,410	3,980
10/3/69	16,630	12,790	3,840
17/3/69	11,940	9,050	2,890
24/3/69	14,520	11,170	3,350
31/3/69	15,650	12,080	3,570
7/4/69	--	--	--
14/4/69	9,720	7,440	2,280
21/4/69	10,510	8,020	2,490
28/4/69	16,130	12,570	3,560
5/5/69	--	--	--
12/5/69	18,620	14,810	3,810
19/5/69	11,210	8,300	2,910
26/5/69	17,870	14,210	3,660

Table B9. Dissolved solids, manure slurry

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	--	--	--
10/2/69	4,950	2,700	2,250
17/2/69	2,680	1,290	1,390
24/2/69	--	--	--
3/3/69	4,740	2,390	2,350
10/3/69	4,450	2,290	2,160
17/3/69	3,620	1,840	1,780
24/3/69	4,160	2,190	1,970
31/3/69	4,860	2,620	2,240
7/4/69	--	--	--
14/4/69	2,970	1,460	1,510
21/4/69	3,270	1,680	1,590
28/4/69	4,360	2,290	2,070
5/5/69	--	--	--
12/5/69	4,550	2,220	2,330
19/5/69	4,020	2,020	2,000
26/5/69	3,920	1,940	1,980

Table B10. Suspended solids, manure slurry

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	--	--	--
10/2/69	12,810	11,130	1,680
17/2/69	10,770	--	--
24/2/69	--	--	--
3/3/69	11,650	10,020	1,630
10/3/69	12,180	10,500	1,680
17/3/69	8,320	7,210	1,110
24/3/69	10,360	8,980	1,380
31/3/69	10,790	9,460	1,330
7/4/69	--	--	--
14/4/69	6,750	5,980	770
21/4/69	7,240	6,340	900
28/4/69	11,770	10,280	1,490
5/5/69	--	--	--
12/5/69	14,070	12,590	1,480
19/5/69	7,190	6,280	910
26/5/69	13,950	12,270	1,680

Table B11. Total solids, lagoon

Date Day/Mo/Yr	Total mg/1	Volatile mg/1	Fixed mg/1
3/2/69	2,250	1,080	1,170
10/2/69	2,340	1,320	1,020
17/2/69	2,510	--	--
24/2/69	--	--	--
3/3/69	2,540	1,170	1,370
10/3/69	2,600	1,170	1,[illegible]30
17/3/69	2,160	1,150	1,010
24/3/69	1,770	760	1,010
31/3/69	1,930	830	1,100
7/4/69	--	--	--
14/4/69	1,950	870	1,080
21/4/69	1,850	780	1,070
28/4/69	1,960	960	1,000
5/5/69	--	--	--
12/5/69	1,840	710	1,130
19/5/69	1,760	580	1,180
26/5/69	2,540	1,090	1,450

Table B12. Dissolved solids, lagoon

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	1,820	--	--
10/2/69	2,040	--	--
17/2/69	2,290	900	1,390
24/2/69	--	--	--
3/3/69	2,460	970	1,490
10/3/69	2,410	890	1,520
17/3/69	1,760	750	1,010
24/3/69	1,500	580	920
31/3/69	1,960	750	1,210
7/4/69	--	--	--
14/4/69	1,600	580	1,020
21/4/69	1,500	510	990
28/4/69	1,440	490	950
5/5/69	--	--	--
12/5/69	1,440	470	970
19/5/69	1,590	460	1,130
26/5/69	1,040	190	1,050

Table B13. Suspended solids, lagoon

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	430	--	--
10/2/69	300	--	--
17/2/69	220	--	--
24/2/69	--	--	--
3/3/69	80	--	--
10/3/69	190	--	--
17/3/69	400	400	0
24/3/69	270	180	90
31/3/69	--	--	--
7/4/69	--	--	--
14/4/69[a]	350	290	60
21/4/69[a]	350	270	80
28/4/69[a]	520	470	50
5/5/69	--	--	--
12/5/69[a]	400	240	160
19/5/69[a]	170	120	50
26/5/69[a]	1,500	900	400

[a] Measured using glassfibre filter.

Table B14. Total solids, returned effluent

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	1,600	610	990
10/2/69	2,310	1,220	1,090
17/2/69	1,890	--	--
24/2/69	--	--	--
3/3/69	1,660	520	1,140
10/3/69	1,620	430	1,190
17/3/69	1,780	630	1,150
24/3/69	1,580	540	1,040
31/3/69	1,520	500	1,020
7/4/69	--	--	--
14/4/69	1,510	470	1,040
21/4/69	1,360	370	990
28/4/69	2,140	580	1,560
5/5/69	--	--	--
12/5/69	2,290	730	1,560
19/5/69	2,120	730	1,390
26/5/69	3,680	2,080	1,600

Table B15. Dissolved solids, returned effluent

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	1,390	--	--
10/2/69	1,380	--	--
17/2/69	1,660	520	1,140
24/2/69	--	--	--
3/3/69	1,640	480	1,160
10/3/69	--	--	--
17/3/69	1,630	550	1,080
24/3/69	1,300	400	900
31/3/69	1,480	450	1,030
7/4/69	--	--	--
14/4/69	1,400	420	980
21/4/69	1,255	305	950
28/4/69	1,795	310	1,485
5/5/69	--	--	--
12/5/69	1,900	420	1,480
19/5/69	1,820	480	1,340
26/5/69	1,640	470	1,170

Table B16. Suspended solids, returned effluent

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	210	--	--
10/2/69	930	--	--
17/2/69	230	--	--
24/2/69	--	--	--
3/3/69	20	--	--
10/3/69	--	--	--
17/3/69	150	80	70
24/3/69	280	140	140
31/3/69	40	--	--
7/4/69	--	--	--
14/4/69[a]	110	50	60
21/4/69[a]	105	65	40
28/4/69[a]	345	270	75
5/5/69	--	--	--
12/5/69[a]	390	310	80
19/5/69[a]	300	250	50
26/5/69[a]	2,040	1,610	430

[a] Measured using glassfibre filter.

Table B17. Total solids, ditch

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	2,480	1,230	1,250
10/2/69	2,960	1,710	1,250
17/2/69	3,390	--	--
24/2/69	--	--	--
3/3/69	4,130	2,300	1,830
10/3/69	3,990	2,220	1,770
17/3/69	4,700	2,930	1,770
24/3/69	4,460	2,780	1,680
31/3/69	4,770	3,080	1,690
7/4/69	--	--	--
14/4/69	5,900	3,800	2,100
21/4/69	4,770	2,910	1,860
28/4/69	5,680	3,580	2,100
5/5/69	--	--	--
12/5/69	5,960	3,850	2,110
19/5/69	5,570	3,620	1,950
26/5/69	4,150	2,450	1,700

Table B18. Dissolved solids, ditch

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	1,350	220	1,130
10/2/69	1,430	220	1,210
17/2/69	1,710	480	1,230
24/2/69	--	--	--
3/3/69	1,720	510	1,210
10/3/69	1,820	530	1,290
17/3/69	1,880	610	1,270
24/3/69	1,470	360	1,110
31/3/69	1,600	450	1,150
7/4/69	--	--	--
14/4/69	1,620	480	1,140
21/4/69	1,430	350	1,080
28/4/69	2,110	530	1,580
5/5/69	--	--	--
12/5/69	2,140	540	1,600
19/5/69	1,920	470	1,450
26/5/69	1,850	580	1,270

Table B19. Suspended solids, ditch

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l	Percent volatile %	Sludge volume index ml/g
3/2/69	1,130	1,010	120	89.4	31
10/2/69	1,530	1,490	40	97.4	29
17/2/69	1,680	--	--	--	30
24/2/69	--	--	--	--	--
3/3/69	2,410	1,790	620	74.3	37
10/3/69	2,170	1,690	480	77.9	41
17/3/69	2,820	2,320	500	82.3	43
24/3/69	2,990	2,420	570	80.9	40
31/3/69	3,170	2,630	540	83	49
7/4/69	--	--	--	--	--
14/4/69	4,280	3,320	960	77.6	44
21/4/69	3,340	2,560	780	76.6	57
28/4/69	3,570	3,050	520	85.4	66
5/5/69	--	--	--	--	--
12/5/69	3,820	3,310	510	86.6	63
19/5/69	3,650	3,150	500	86.3	59
26/5/69	2,300	1,870	430	81.3	52

Table B20. Linear regression for volatile suspended solids, ditch

Date Day/Mo/Yr	X Day	Y VSS mg/1
3/2/69	0	1,010
10/2/69	7	1,490
17/2/69	14	--
24/2/69	21	--
3/3/69	28	1,790
10/3/69	35	1,690
17/3/69	42	2,320
24/3/69	49	2,420
31/3/69	56	2,630
7/4/69	63	--
14/4/69	70	3,320

$\Sigma X = 287$ $\Sigma Y = 16{,}670$ $(\Sigma X)^2/8 = 10{,}296$

$\Sigma X^2 = 14{,}259$ $\Sigma XY = 715{,}400$ $(\Sigma X)(\Sigma Y)/8 = 598{,}036$

$$\text{Growth rate} = \frac{\Sigma XY - (\Sigma X)(\Sigma Y)/8}{\Sigma X^2 - (\Sigma X)^2/8} = 29.6 \text{ mg/1 day}$$

or 11.6 lb/day

Table B21. Total solids, sludge

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	7,800	5,400	2,400
10/2/69	27,600	20,440	7,160
17/2/69	30,120	--	--
24/2/69	--	--	--
3/3/69	17,990	13,070	4,920
10/3/69	18,350	13,790	4,560
17/3/69	19,880	14,400	5,480
24/3/69	20,410	15,230	5,180
31/3/69	14,270	10,570	3,700
7/4/69	--	--	--
14/4/69	17,880	12,820	5,060
21/4/69	14,930	10,430	4,500
28/4/69	18,930	13,950	4,980
5/5/69	--	--	--
12/5/69	18,370	14,420	3,950
19/5/69	18,690	14,600	4,090
26/5/69	--	--	--

Table B22. Dissolved solids, sludge

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
3/2/69	1,850	400	1,450
10/2/69	1,500	550	950
17/2/69	1,680	520	1,160
24/2/69	--	--	--
3/3/69	1,750	530	1,220
10/3/69	1,810	460	1,350
17/3/69	1,810	560	1,250
24/3/69	1,430	380	1,050
31/3/69	1,540	420	1,120
7/4/69	--	--	--
14/4/69	1,490	420	1,070
21/4/69	1,310	290	1,020
28/4/69	2,210	580	1,630
5/5/69	--	--	--
12/5/69	2,100	520	1,580
19/5/69	1,910	480	1,430
26/5/69	--	--	--

Table B23. Suspended solids, sludge

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l	Percent volatile %
3/2/69	5,950	5,000	950	84
10/2/69	26,100	19,890	6,210	76.2
17/2/69	28,440	--	--	--
24/2/69	--	--	--	--
3/3/69	16,240	12,540	3,700	77.2
10/3/69	16,540	13,330	3,210	80.6
17/3/69	18,070	13,840	4,230	76.6
24/3/69	18,980	14,850	4,130	78.2
31/3/69	12,730	10,150	2,580	79.7
7/4/69	--	--	--	--
14/4/69	16,390	12,400	3,990	75.7
21/4/69	13,620	10,140	3,480	74.4
28/4/69	16,720	13,370	3,350	80
5/5/69	--	--	--	--
12/5/69	16,270	13,900	2,370	85.4
19/5/69	16,780	14,120	2,660	84.1
26/5/69	--	--	--	--

Table B24. Oxygen demand, manure slurry

Date Day/Mo/Yr	COD mg/l	BOD_5 mg/l
3/2/69	7,080	2,830
10/2/69	13,500	4,770
17/2/69	17,600	4,810
24/2/69	--	--
3/3/69	--	6,750
10/3/69	22,700	8,280
17/3/69	17,800	8,800
24/3/69	17,100	7,850
31/3/69	22,600	8,700
7/4/69	27,200	--
14/4/69	22,300	10,200
21/4/69	23,800	10,000
28/4/69	14,900	8,500
5/5/69	--	--
12/5/69	25,000	9,800
19/5/69	23,700	8,700
26/5/69	20,600	9,100

Table B25. Oxygen demand, lagoon

Date Day/Mo/Yr	COD mg/l	BOD_5 mg/l
3/2/69	831	728
10/2/69	--	890
17/2/69	--	1,120
24/2/69	--	--
3/3/69	2,420	1,080
10/3/69	2,500	1,240
17/3/69	1,670	765
24/3/69	1,560	960
31/3/69	2,270	885
7/4/69	1,820	600
14/4/69	1,940	1,260
21/4/69	2,140	1,230
28/4/69	1,800	1,030
5/5/69	--	--
12/5/69	1,290	375
19/5/69	960	345
26/5/69	1,050	276

Table B26. Oxygen demand, returned effluent

Date Day/Mo/Yr	COD mg/l	BOD_5 mg/l
3/2/69	425	175
10/2/69	570	95
17/2/69	790	139
24/2/69	--	--
3/3/69	557	67
10/3/69	440	116
17/3/69	452	71
24/3/69	--	475
31/3/69	544	159
7/4/69	408	80
14/4/69	480	119
21/4/69	425	90
28/4/69	1,420	465
5/5/69	--	--
12/5/69	990	170
19/5/69	--	2,050
26/5/69	2,810	1,340

Table B27. Air temperature, pig population, flow rate and power

Date Day/Mo/Yr	Outside Unit K °F	Inside Unit K °F	Pig population	Tank discharge gpw	Rotor power kwh/wk
3/2/69	35.5	45	640	22,500	--
10/2/69	20	48.5	640	31,000	--
17/2/69	35	60	640	37,000	--
24/2/69	--	--	640	20,500	434
3/3/69	26	54.5	661	26,000	434
10/3/69	37	57	661	46,697	482
17/3/69	43	63	661	28,209	417
24/3/69	38.5	65	661	28,588	412
31/3/69	46	65	661	28,457	417
7/4/69	65	73	618	23,281	333
14/4/69	54	67	618	28,393	461
21/4/69	68	73.5	618	27,682	370
28/4/69	62	74	618	25,806	388
5/5/69	80	79	666	10,160	--
12/5/69	84	84	666	15,576	--
19/5/69	62	71	666	33,362	368
26/5/69	83	83	666	25,892	346

APPENDIX C. RAW RESULTS PHASE 2

Table C1. Nitrogen forms, manure slurry

Date Day/Mo/Yr	Kjeldahl mg/l	Ammonia mg/l	Organic mg/l	Nitrate mg/l
24/8/70	670	314	356	0
31/8/70	754	510	244	0
7/9/70	650	260	390	0
14/9/70	--	--	--	--
21/9/70	443	130	313	95
28/9/70	1,220	385	835	40
5/10/70	1,280	427	853	3
12/10/70	1,340	361	979	38
28/12/70	1,240	313	827	0
4/12/71	--	--	--	--
11/1/71	1,220	340	780	140
18/1/71	1,690	382	1,308	255
25/1/71	2,015	502	1,513	210
1/2/71	2,570	506	2,064	60
8/2/71	2,650	580	2,070	0
15/2/71	1,870	362	1,508	40
22/2/71	2,380	490	1,890	0
1/3/71	2,360	481	1,879	0

Table C2. Nitrogen forms, returned effluent

Date Day/Mo/Yr	Kjeldahl mg/l	Ammonia mg/l	Organic mg/l	Nitrate mg/l
24/8/70	615	187	428	0
31/8/70	233	174	59	0
7/9/70	450	249	201	0
14/9/70	--	--	--	--
21/9/70	22.5	5.5	17	90
28/9/70	53	6	47	60
5/10/70	56	1.5	54.5	10
12/10/70	32	1.5	30.5	60
28/12/70	380	46	336	10
4/12/70	--	--	--	--
11/1/71	431	76	355	215
18/1/71	974	185	789	335
25/1/71	1,060	218	842	280
1/2/71	600	154	446	150
8/2/71	1,210	118	1,092	0
15/2/71	1,140	Trace	1,140	100
22/2/71	1,030	0	1,030	12
1/3/71	1,170	13	1,157	0

Table C3. Nitrogen forms and pH, ditch

Date Day/Mo/Yr	Kjeldahl mg/l	Ammonia mg/l	Organic mg/l	Nitrate mg/l	pH
11/1/71	1,240	76	1,164	225	--
18/1/71	1,050	185	865	315	7.0
25/1/71	1,460	276	1,184	260	7.3
1/2/71	1,280	173	1,107	170	7.5
8/2/71	1,390	123	1,267	0	7.7
15/2/71	1,340	Trace	1,340	130	7.2
22/2/71	1,140	1.5	1,138	40	7.6
1/3/71	1,460	14	1,446	0	7.6

Table C4. Nitrogen forms, over flow

Date Day/Mo/Yr	Kjeldahl mg/l	Ammonia mg/l	Organic mg/l
11/1/71	734	76	658
18/1/71	442	171	271
25/1/71	760	233	527
1/2/71	1,320	190	1,130
8/2/71	830	106	726
15/2/71	1,620	8	1,612
22/2/71	526	4	522
1/3/71	--	48	--

Table C5. Phosphate (as PO_4^{-3}), manure slurry

Date Day/Mo/Yr	Ortho mg/l	Poly mg/l	Total mg/l
24/8/70	--	550	450
31/8/70	--	1,100	715
7/9/70	--	1,150	1,220
14/9/70	--	--	--
21/9/70	670	1,300	770
28/9/70	--	1,000	1,000
5/10/70	1,000	1,490	1,300
12/10/70	1,390	1,430	1,920
28/12/70	850	1,020	1,420
4/12/71	--	--	--
11/1/71	1,000	1,200	1,250
18/1/71	1,540	1,490	1,980
25/1/71	1,590	1,000	1,520
1/2/71	900	900	850
8/2/71	1,870	1,870	2,050
15/2/71	1,810	1,750	1,870
22/2/71	1,490	1,540	1,920
1/3/71	1,590	1,650	1,980

Table C6. Phosphate (as PO_4^{-3}), returned effluent

Date Day/Mo/Yr	Ortho mg/l	Poly mg/l	Total mg/l
24/8/70	--	450	700
31/8/70	--	215	215
7/9/70	--	1,380	1,510
14/9/70	--	--	--
21/9/70	345	500	385
28/9/70	--	500	500
5/10/70	430	1,340	420
12/10/70	625	430	430
28/12/70	500	820	820
4/12/71	--	--	--
11/1/71	660	930	580
18/1/71	930	1,000	1,390
25/1/71	1,000	1,750	1,000
1/2/71	960	960	490
8/2/71	1,250	1,390	1,540
15/2/71	1,300	1,540	1,540
22/2/71	1,100	1,340	1,700
1/3/71	1,340	1,540	1,750

Table C7. pH, chloride, sulphate, temperature and coliforms, manure slurry

Date Day/Mo/Yr	pH	Cl^- mg/l	SO_4^{-2} mg/l	Temp °F	Coliforms[a] N°/100 ml
24/8/70	7.7	--	--	76	1.1×10^7
31/8/70	7.6	235	275	78	0.87×10^7
7/9/70	--	205	--	73	--
14/9/70	--	--	--	--	--
21/9/70	7.1	240	240	69.5	11×10^7
28/9/70	7.2	470	365	62	19×10^7
5/10/70	7.2	527	345	63	12×10^7
12/10/70	7.0	474	--	64	--
28/12/70	7.0	465	--	45	--
4/12/71	--	--	--	--	--
11/1/71	6.9	267	430	49	6.3×10^7
18/1/71	6.9	720	440	47.5	6.0×10^7
25/1/71	7.3	680	580	50.5	--
1/2/71	7.2	860	940	50	6.2×10^7
8/2/71	7.6	780	650	53	3.4×10^7
15/2/71	7.5	630	560	--	5.5×10^7
22/3/71	7.4	760	590	53.5	6.5×10^7
1/3/71	7.4	830	560	55	2.6×10^7

[a] Membrane filter technique.

Table C8. pH, chloride, sulphate, temperature and coliforms, returned effluent

Date Day/Mo/Yr	pH	Cl^- mg/l	SO_4 mg/l	Temp °F	Coliforms[a] N°/100 ml
24/8/70	7.8	115	198	79.5	2.0×10^6
31/8/70	7.7	152	180	82	0.31×10^6
7/9/70	--	199	--	79	--
14/9/70	--	--	--	--	--
21/9/70	6.8	168	240	74	9.5×10^6
28/9/70	7.0	216	325	71.5	2.4×10^6
5/10/70	7.2	270	335	70.5	1.2×10^6
12/10/70	6.9	315	--	69	--
28/12/70	7.0	265	420	45.5	--
4/12/71	--	--	--	--	--
11/1/71	6.4	267	355	45	1.1×10^6
18/1/71	6.8	335	380	41	12×10^6
25/1/71	7.2	400	440	47	--
1/2/71	7.6	443	570	46.5	2.7×10^6
8/2/71	7.6	520	550	48.5	2.1×10^6
15/2/71	7.2	570	470	52.5	3.0×10^6
22/2/71	7.5	540	610	50.5	3.0×10^6
1/3/71	7.5	540	500	50.5	2.1×10^6

[a] Membrane filter technique.

Table C9. Total solids, manure slurry

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
24/8/70	11,600	8,460	3,140
31/8/70	12,720	9,070	3,650
7.9/70	7,760	5,920	1,840
14/9/70	--	--	--
21/9/70	5,360	3,250	2,110
28/9/70	11,610	7,030	4,580
5/10/70	11,930	8,500	3,430
12/10/70	7,820	4,590	3,230
28/12/70	16,810	13,290	3,520
4/12/71	--	--	--
11/1/71	14,500	10,980	3,520
18/1/71	17,650	13,010	4,640
25/1/71	23,440	17,870	5,570
1/2/71	20,490	15,260	5,230
8/2/71	25,960	19,990	5,970
15/2/71	39,240	31,500	7,740
22/3/71	26,100	19,920	6,180
1/3/71	29,560	23,120	6,440

Table C10. Dissolved solids, manure slurry

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
24/8/70	2,490	1,300	1,190
31/8/70	6,350	3,790	2,560
7/9/70	2,300	1,160	1,140
14/9/70	--	--	--
21/9/70	2,910	1,240	1,670
28/9/70	5,350	2,750	2,600
5/10/70	4,870	2,490	2,380
12/10/70	4,620	1,920	2,700
28/12/70	4,560	2,890	1,670
4/12/71	--	--	--
11/1/71	7.610	5,120	2,490
18/1/71	6,990	4,000	2,990
25/1/71	7,010	3,870	3,140
1/2/71	4,860	2,030	2,830
8/2/71	3,640	840	2,800
15/2/71	6,100	2,740	3,360
22/2/71	5,860	2,680	3,180
1/3/71	7,320	3,620	3,700

Table C11. Suspended solids, manure slurry

Date Day/Mo/Yr	Total mg/1	Volatile mg/1	Fixed mg/1
24/8/70	9,100	7,160	1,950
31/8/70	6,370	5,280	1,090
7/9/70	5,460	4,760	700
14/9/70	--	--	--
21/9/70	2,450	2,010	440
28/9/70	6,260	4,280	1,980
5/10/70	7.060	6,010	1,050
12/10/70	3,200	2,670	530
28/12/70	12,250	10,400	1,850
4/12/71	--	--	--
11/1/71	6,890	5,860	1,030
18/1/71	10,660	9,010	1,650
25/1/71	16,430	14,000	2,430
1/2/71	15,630	13,230	2,400
8/2/71	22,320	19,150	3,170
15/2/71	33,140	28,760	4,380
22/2/71	20,240	17,240	3,000
1/3/71	22,240	19,500	2,740

Table C12. Total solids, returned effluent

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
24/8/70	7,100	4,500	2,600
31/8/70	1,850	840	1,010
7/9/70	1,660	740	920
14/9/70	--	--	--
21/9/70	2,740	1,230	1,510
28/9/70	2,390	770	1,620
5/10/70	2,420	710	1,710
12/10/70	2,630	610	2,020
28/12/70	15,910	12,410	3,500
4/12/71	--	--	--
11/1/71	4,210	2,070	2,140
18/1/71	6,970	4,320	2,650
25/1/71	15,430	11,210	4,220
1/2/71	16,710	11,900	4,810
8/2/71	15,670	10,920	4,750
15/2/71	17,770	12,440	5,330
22/2/71	17,490	12,270	5,220
1/3/71	16,370	11,140	5,230

Table C13. Dissolved solids, returned effluent

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
24/8/70	900	230	670
31/8/70	1,240	320	920
7/9/70	1,190	320	870
14/9/70	--	--	--
21/9/70	1,650	370	1,280
28/9/70	1,910	460	1,450
5/10/70	2,330	640	1,690
12/10/70	2,450	480	1,970
28/12/70	2,050	540	1,510
4/12/71	--	--	--
11/1/71	3,910	1,640	2,270
18/1/71	4,240	2,180	2,060
25/1/71	3,900	1,510	2,390
1/2/71	3,670	1,030	2,640
8/2/71	5,900	1,500	4,400
15/2/71	3,960	940	3,020
22/2/71	3,260	650	2,610
1/3/71	3,430	800	2,630

Table C14. Suspended solids, returned effluent

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
24/8/70	6,200	4,270	1,930
31/8/70	610	520	90
7/9/70	470	420	50
14/9/70	--	--	--
21/9/70	1,090	860	230
28/9/70	480	310	170
5/10/70	90	70	20
12/10/70	180	130	50
28/12/70	13,860	11,870	1,990
4/12/71	--	--	--
11/1/71	300	430	--
18/1/71	2,730	2,140	590
25/1/71	11,530	9,700	1,830
1/2/71	13,040	10,870	2,170
8/2/71	9,770	9,420	350
15/2/71	13,810	11,500	2,310
22/2/71	14,230	11,620	2,610
1/3/71	12,940	10,340	2,600

Table C15. Total solids, ditch

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
24/8/70	6,620	4,190	2,430
31/8/70	9,040	6,010	3,030
7/9/70	8,750	6,010	2,740
14/9/70	--	--	--
21/9/70	7,800	4,190	3,610
28/9/70	8,680	5,650	3,030
5/10/70	9,930	6,550	3,380
12/10/70	11,500	7,530	3,970
28/12/70	13,730	10,660	3,070
4/12/71	--	--	--
11/1/71	16,250	12,560	3,690
18/1/71	16,570	12,410	4,160
25/1/71	19,860	15,050	4,810
1/2/71	19,750	14,690	5,060
8/2/71	17,250	12,250	5,000
15/2/71	21,410	15,670	5,740
22/2/71	18,420	13,040	5,380
1/3/71	18,340	12,850	5,490

Table C16. Dissolved solids, ditch

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
24/8/70	820	140	680
31/8/70	1,300	370	930
7/9/70	1,190	310	880
14/9/70	--	--	--
21/9/70	1,720	420	1,300
28/9/70	2,100	530	1,570
5/10/70	2,300	600	1,700
12/10/70	2,430	470	1,960
28/12/70	2,210	710	1,500
4/12/71	--	--	--
11/1/71	3,990	1,710	2,280
18/1/71	4,220	2,140	2,080
25/1/71	4,170	1,670	2,500
1/2/71	3,750	1,100	2,650
8/2/71	4,030	1,000	3,030
15/2/71	4,150	1,010	3,140
22/2/71	3,300	640	2,660
1/3/71	3,280	920	2,360

Table C17. Suspended solids, ditch

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l	Percent volatile %	Sludge volume index ml/g
24/8/70	5,800	4,050	1,750	69.8	58
31/8/70	7,740	5,640	2,100	72.9	67
7/9/70	7,560	5,700	1,860	75.4	72
14/9/70	--	--	--	--	--
21/9/70	6,080	3,770	2,310	62.0	91
28/9/70	6,580	5,120	1,460	77.8	74
5/10/70	7,630	5,950	1,680	78.0	79
12/10/70	9,070	7,060	2,010	77.8	108
28/12/70	11,520	9,950	1,570	86.4	65
4/12/71	--	--	--	--	--
11/1/71	12,260	10,850	1,410	88.5	74
18/1/71	12,350	10,270	2,080	83.2	80
25/1/71	15,690	13,380	2,310	85.3	62
1/2/71	16,000	13,590	2,410	84.9	62
8/2/71	13,220	11,250	1,970	85.1	--
15/2/71	17,260	14,660	2,600	84.9	--
22/2/71	15,120	12,400	2,720	82.0	--
1/3/71	15,060	11,930	3,130	79.2	--

Table C18. Total solids, sludge

Date Day/Mo/Yr	Total mg/1	Volatile mg/1	Fixed mg/1
24/8/70	20,230	12,830	7,400
31/8/70	21,770	15,250	6,520
7/9/70	17,490	12,670	4,820
14/9/70	--	--	--
21/9/70	11,610	8,360	3,250
28/9/70	7,900	5,030	2,870
5/10/70	12,260	7,290	4,970
12/10/70	14,430	9,960	4,470
28/12/70	29,440	24,850	4,590
4/12/71	--	--	--
11/1/71	16,520	12,710	3,810
18/1/71	15,520	11,500	4,020
25/1/71	--	--	--
1/2/71	19,820	14,610	5,210
8/2/71	17,270	12,270	5,000
15/2/71	18,750	13,360	5,390
22/2/71	18,480	13,020	5,460
1/3/71	16,800	11,500	5,300

Table C19. Dissolved solids, sludge

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
24/8/70	950	230	720
31/8/70	1,470	460	1,010
7/9/70	1,240	310	930
14/9/70	--	--	--
21/9/70	1,620	400	1,220
28/9/70	2,080	510	1,570
5/10/70	2,250	540	1,710
12/10/70	2,460	450	2,010
28/12/70	2,130	580	1,550
4/12/71	--	--	--
11/1/71	3,920	1,560	2,360
18/1/71	5,310	3,180	2,130
25/1/71	--	--	--
1/2/71	3,710	1,160	2,550
8/2/71	3,900	940	2,960
15/2/71	4,130	1,000	3,130
22/2/71	3,390	660	2,730
1/3/71	3,570	700	2,870

Table C20. Suspended solids, sludge

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l	Percent volatile %
24/8/70	19,280	12,600	6,680	65.4
31/8/70	20,300	14,790	5,510	72.9
7/9/70	16,250	12,360	3,890	76.1
14/9/70	--	--	--	--
21/9/70	9,990	7,960	2,030	79.7
28/9/70	5,820	4,520	1,300	77.7
5/10/70	10,010	6,750	3,260	67.4
12/10/70	11,970	9,510	2,460	79.4
28/12/70	27,310	24,270	3,040	88.9
4/12/71	--	--	--	--
11/1/71	12,600	11,150	1,450	88.5
18/1/71	10,210	8,320	1,890	81.5
25/1/71	--	--	--	--
1/2/71	16,110	13,450	2,660	83.5
8/2/71	13,370	11,330	2,040	84.7
15/2/71	14,620	12,360	2,260	84.5
22/2/71	15,090	12,360	2,730	81.9
1/3/71	13,230	10,800	2,430	81.6

Table C21. Total solids, overflow

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
11/1/71	11,170	8,140	3,030
18/1/71	6,290	3,820	2,470
25/1/71	26,100	20,480	5,620
1/2/71	16,410	12,190	4,220
8/2/71	15,270	10,600	4,670
15/2/71	18,110	12,750	5,360
22/2/71	15,560	10,660	4,900
1/3/71	19,750	14,160	5,590

Table C22. Suspended solids, overflow

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
11/1/71	7,260	6,500	760
18/1/71	2,050	1,640	410
25/1/71	22,200	18,970	3,230
1/2/71	12,740	11,160	1,580
8/2/71	9,370	9,100	270
15/2/71	14,150	11,810	2,340
22/2/71	12,300	10,010	2,290
1/3/71	16,320	13,360	2,960

Table C23. Oxygen demand/manure slurry

Date Day/Mo/Yr	COD mg/1	BOD_5 mg/1	BOD_5 + N-Serve[a] mg/1
24/8/70	6,550	2,950	--
31/8/70	6,850	2,900	--
7/9/70	9,800	3,850	--
14/9/70	--	--	--
21/9/70	10,350	4,130	--
28/9/70	17,600	8,300	--
5/10/70	10,000	5,550	3,700
12/10/70	--	--	--
28/12/70	16,000	6,900	--
4/12/71	--	--	--
11/1/71	16,100	7,530	6,000
18/1/71	20,800	8,100	5,250
25/1/71	30,800	8,050	6,180
1/2/71	32,900	16,800	8,730
8/2/71	26,300	10,800	7,450
15/2/71	27,700	11,300	8,030
22/2/71	31,700	13,800	10,300
1/3/71	27,700	16,200	9,680

[a] Registered trademark of the Dow Chemical Company.

Table C24. Oxygen demand/returned effluent

Date Day/Mo/Yr	COD mg/l	BOD_5 mg/l	BOD_5 + N-Serve[a] mg/l	Centrate COD mg/l	Centrate BOD_5 mg/l
24/8/70	530	141	--	--	--
31/8/70	1,630	495	--	--	--
7/9/70	840	134	--	--	--
14/9/70	--	--	--	--	--
21/9/70	490	40	--	--	--
28/9/70	860	126	--	--	--
5/10/70	750	69	60	--	--
12/10/70	--	--	--	--	--
28/12/70	510	58	--	--	--
4/12/71	--	--	--	--	--
11/1/71	8,200	2,550	1,140	--	--
18/1/71	--	4,220	2,470	--	--
25/1/71	14,200	3,920	2,570	1,030	72
1/2/71	18,700	4,430	2,910	1,010	74
8/2/71	15,900	4,050	2,850	880	54
15/2/71	18,400	5,950	3,970	740	45
22/2/71	17,600	6,450	4,100	1,110	240
1/3/71	14,700	5,430	2,910	726	45

[a] Registered trademark of the Dow Chemical Company.

Table C25. Oxygen demand, ditch

Date Day/Mo/Yr	COD mg/l	BOD_5 mg/l	Centrate COD mg/l	 BOD_5 mg/l
11/1/71	15,100	4,400	--	--
18/1/71	18,600	5,600	2,800	--
25/1/71	21,600	5,550	965	66
1/2/71	20,400	5,800	945	42
8/2/71	21,000	7,050	900	60
15/2/71	21,500	6,900	720	33
22/2/71	19,500	7,300	920	138
1/3/71	18,700	6,550	772	57

Table C26. Oxygen demand, overflow

Date Day/Mo/Yr	COD mg/l	BOD_5 mg/l
11/1/71	9,800	2,100
18/1/71	--	1,540
25/1/71	18,600	3,730
1/2/71	14,900	4,180
8/2/71	10,800	2,770
15/2/71	17,600	5,300
22/2/71	11,700	4,140
1/3/71	9,450	3,420

Table C27. Air temperature and relative humidity

Date	Outside Unit K		Inside Unit K	
Day/Mo/Yr	Temp °F	RH %	Temp °F	RH %
24/8/70	67.5	79.5	75	87
31/8/70	67	75	68	84
7/9/70	71	69	74	76
14/9/70	--	--	--	--
21/9/70	68	67	68.5	69
28/9/70	57	52	59.5	66
5/10/70	62	65	64	73.5
12/10/70	59.5	64.5	65.5	64.5
28/12/70	29	42.5	54.5	93.5
4/12/71	18	49	59	97
11/1/71	17	45	59.5	94
18/1/71	17.5	29	60	92
25/1/71	18	46.5	62.5	91
1/2/71	14	49	62	93
8/2/71	15.5	50.5	63	92
15/2/71	38.5	72	66	92
22/2/71	31.5	64.5	64	87
1/3/71	30.5	57	60	94

Table C28. Pig population, flow rate and power

Date Day/Mo/Yr	Pig population	Tank discharge gpw	Ditch overflow gpw	Rotor power kwh/wk	Total power kwh/wk
24/8/70	575	31,620	--	1,040	--
31/8/70	575	27,534	--	915	--
7/9/70	496	22,863	--	--	1,124
14/9/70	496	22,273	--	--	1,126
21/9/70	496	32,980	--	--	1,162
28/9/70	496	10,587[a]	--	1,028	--
5/10/70	446	12,137[b]	--	--	1,192
12/10/70	468	16,120	--	1,122	--
28/12/70	575	--	--	--	--
4/12/71	595	--	5,653	950	--
11/1/71	582	20,680	--	1,046	--
18/1/71	616	19,900	4,011	--	1,225
25/1/71	632	19,380	5,091	1,018	--
1/2/71	621	16,820	8,125	867	--
8/2/71	596	13,580	5,077	1,065	--
15/2/71	613	15,120	14,983	--	1,124
22/2/71	626	15,840	4,551	678	--
1/3/71	662	14,100	4,177	--	1,204

[a] 3 hr flush interval.

[b] 2 hr flush interval from here on.

APPENDIX D. RAW RESULTS AEROBIC SLUDGE DIGESTION

Table D1. Nitrogen forms, chloride and temperature

Date Day/Mo/Yr	Kjeldahl mixed mg/1	Kjeldahl centrate mg/1	Nitrate mg/1	Chloride mg/1	Temp °F
8/3/71	1,460	41	65	710	50.5
15/3/71	1,370	41	--	714	50
22/3/71	1,195	45	--	715	47
29/3/71	927	40	33	700	49.5
5/4/71	734	34	--	694	52
12/4/71	790	34	--	717	57
19/4/71	613	28	--	729	64.5
26/4/71	593	24	270	729	60
3/5/71	504	21	240	724	59.5
10/5/71	476	17	60	725	62
17/5/71	454	22	45	724	67

Table D2. Total solids

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
8/3/71	21,780	15,350	6,430
15/3/71	19,790	13,530	6,260
22/3/71	18,180	11,970	6,210
29/3/71	16,010	10,400	5,610
5/4/71	13,700	8,500	5,200
12/4/71	15,920	10,290	5,630
19/4/71	14,270	9,100	5,170
26/4/71	13,610	8,370	5,240
3/5/71	11,300	6,340	4,960
10/5/71	11,240	6,360	4,880
17/5/71	11,790	6,740	5,050

Table D3. Dissolved solids

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l
8/3/71	4,370	990	3,380
15/3/71	4,430	1,000	3,430
22/3/71	4,420	890	3,530
29/3/71	4,190	990	3,200
5/4/71	4,650	870	3,780
12/4/71	4,040	650	3,390
19/4/71	4,150	760	3,390
26/4/71	3,970	690	3,280
3/5/71	3,980	650	3,330
10/5/71	3,910	640	3,270
17/5/71	4,050	770	3,280

Table D4. Suspended solids

Date Day/Mo/Yr	Total mg/l	Volatile mg/l	Fixed mg/l	Percent volatile	Sludge volume index ml/g
8/3/71	17,410	14,360	3,050	82.5	NS[a]
15/3/71	15,360	12,530	2,830	81.6	NS
22/3/71	13,760	11,080	2,680	80.5	72
29/3/71	11,820	9,410	2,410	79.6	NS
5/4/71	9,050	7,630	1,420	84.3	109
12/4/71	11,880	9,640	2,240	81.1	78
19/4/71	10,120	8,340	1,780	82.4	65
26/4/71	9,640	7,680	1,960	79.7	51
3/5/71	7,320	5,690	1,630	77.7	74
10/5/71	7,330	5,720	1,610	78.0	72
17/5/71	7,740	5,970	1,770	77.1	114

[a] NS = no settling.

Table D5. Oxygen demand, rotor power and temperature

Date Day/Mo/Yr	Mixed mg/l	Centrate COD mg/l	Rotor power kwh	Temp °F
8/3/71	23,800	1,240	1,257	34
15/3/71	19,100	1,060	1,251	36
22/3/71	19,400	915	1,253	32
29/3/71	14,000	856	1,216	45
5/4/71	12,900	760	1,171	59.5
12/4/71	14,610	703	1,137	54
19/4/71	16,400	657	1,118	61
26/4/71	10,900	581	1,129	55
3/5/71	9,700	547	1,120	66
10/5/71	8,070	440	1,099	63
17/5/71	9,500	406	--	67

Table D6. Linear regression for K rate volatile suspended solids

Date Day/Mo/Yr	X Day	VSS mg/1	Y Log_{10} VSS
8/3/71	0	14,360	4.157
15/3/71	7	12,530	4.098
22/3/71	14	11,080	4.045
29/3/71	21	9,410	3.974
5/4/71	28	7,630	3.883
12/4/71	35	9,640	3.984
19/4/71	42	8,340	3.921
26/4/71	49	7,680	3.885
3/5/71	56	5,690	3.755
10/5/71	63	5,720	3.757
17/5/71	70	5,970	3.776

$\Sigma X = 385$

$\Sigma Y = 43{,}235$

$(\Sigma X)^2/11 = 13{,}475$

$\Sigma X^2 = 18{,}865$

$\Sigma XY = 1{,}483{,}272$

$(\Sigma X)(\Sigma Y)/11 = 1{,}513{,}225$

$$K = -\frac{\Sigma XY - (\Sigma X)(\Sigma Y)/11}{\Sigma X^2 - (\Sigma X)^2/11} = 0.00556 \text{ Day}^{-1}$$

$$\text{Log}_{10}\, VSS_0 = Y_0 = \bar{Y} + K\bar{X}$$

$$= 4.125 \quad \text{or} \quad VSS_0 = 13{,}350 \text{ mg/1}$$

Table D7. Linear regression for K rate COD in suspended solids

Date Day/Mo/Yr	X Day	COD mg/1	Y $\text{Log}_{10}\text{COD}$
8/3/71	0	22,600	4.354
15/3/71	7	18,040	4.256
22/3/71	14	18,490	4.267
29/3/71	21	13,140	4.119
5/4/71	28	12,140	4.084
12/4/71	35	13,910	4.143
19/4/71	42	15,740	4.197
26/4/71	49	10,320	4.014
3/5/71	56	9,150	3.961
10/5/71	63	7,630	3.883
17/5/71	70	9,090	3.959

$\Sigma X = 385$ $\Sigma Y = 45{,}237$ $(\Sigma X)^2/11 = 13{,}475$

$\Sigma X^2 = 18{,}865$ $\Sigma XY = 1{,}551.921$ $(\Sigma X)(\Sigma Y)/11 = 1{,}583{,}295$

$$K = -\frac{\Sigma XY - (\Sigma X \Sigma Y)/11}{\Sigma X^2 - (\Sigma X)^2/11} = 0.00583 \text{ Day}^{-1}$$

$$\text{Log}_{10}\text{COD}_0 = Y_0 = \bar{Y} + K\bar{X}$$

$$= 4.316 \quad \text{or} \quad \text{COD}_0 = 20{,}700 \text{ mg/1}$$

Table D8. Linear regression for K rate Kjeldahl N in suspended solids

Date Day/Mo/Yr	X Day	Kjeldahl N mg/l	Y Log_{10}N
8/3/71	0	1,430	3.152
15/3/71	7	1,330	3.124
22/3/71	14	1,150	3.061
29/3/71	21	887	2.948
5/4/71	28	700	2.845
12/4/71	35	756	2.879
19/4/71	42	685	2.836
26/4/71	49	569	2.755
3/5/71	56	483	2.684
10/5/71	63	459	2.662
17/5/71	70	432	2.635

$\Sigma X = 385$ $\Sigma Y = 31{,}581$ $(\Sigma X)^2/11 = 13{,}475$

$\Sigma X^2 = 18{,}865$ $\Sigma XY = 1{,}063{,}622$ $(\Sigma X)(\Sigma Y)/11 = 1{,}105{,}335$

$$K = -\frac{\Sigma XY - (\Sigma X)(\Sigma Y)/11}{\Sigma X^2 - (\Sigma X)^2/11} = 0.00774 \text{ Day}^{-1}$$

$$\text{Log}_{10} N_0 = Y_0 = \bar{Y} + K\bar{X}$$

$$= 3.142 \quad \text{or} \quad N_0 = 1{,}390 \text{ mg/l}$$

APPENDIX E. RAW RESULTS PIGS' WATER SUPPLY

Table E1. Pig weight gain pen 16 gutter water (G)

Pig	Body weight in lb				Weight gain lb
	5/1/71	19/1/71	2/2/71	10/2/71	
2958 B	46	62	78	84	38
4022 B	46	66	87	102	56
2951 S	57	62	70	75	18
4046 B	53	74	94	116	63
4049 B	60	81	95	105	45
4044 B	61	86	93	103	42
2195 B	68	76	93	101	33
4070 S	52	71	78	86	34
2972 S	54	70	80	90	36
2934 S	67	84	89	103	36

Table E2. Pig weight gain pen 17 normal waterer (N)

Pig	Body weight in lb				Weight gain lb
	5/1/71	19/1/71	2/2/71	10/2/71	
2954 B	69	102	110	127	58
2994 B	56	82	98	122	66
4000 S	61	90	115	128	67
4043 S	59	90	110	121	62
4009 B	62	96	120	138	76
4060 S	53	82	103	112	59
4063 B	51	80	103	117	66
2193 B	73	108	126	142	69
4821 S	56	82	100	112	56
4820 S	48	62	71	85	37

Table E3. Pig weight gain pen 18 gutter water (G)

Pig	Body weight in lb				Weight gain lb
	5/1/71	19/1/71	2/2/71	10/2/71	
4233 B	53	75	98	108	55
4137 B	40	54	71	80	40
4234 B	63	90	117	127	64
4300 S	44	60	79	89	45
4065 B	52	68	85	94	42
4162 B	62	88	113	124	62
4040 B	64	78	101	112	48
4016 B	72	74	87	96	24
2192 B	82	98	124	135	53
2952 S	55	74	95	104	49

Table E4. Pig weight gain pen 19 normal waterers (N)

Pig	Body weight in lb				Weight gain lb
	5/1/71	19/1/71	2/2/71	10/2/71	
4133 S	48	66	97	100	52
4153 S	55	72	83	92	37
4157 S	61	84	107	117	56
4064 S	53	72	90	96	43
4126 B	52	74	100	112	60
4291 S	58	80	104	117	59
4123 S	55	78	103	118	63
4210 B	50	74	106	110	60
4134 B	50	73	92	108	58
4260 S	50	70	99	112	62

Table E5. Pig weight gain pen 20 gutter water (G)

Pig	Body weight in lb				Weight gain lb
	5/1/71	19/1/71	2/2/71	10/2/71	
4090 S	63	80	105	122	59
4315 B	55	74	81	91	36
4160 S	58	74	91	100	42
4100 S	54	70	91	102	48
4101 S	57	69	82	96	39
4232 S	58	74	93	102	44
4116 B	44	52	67	76	32
4220 S	63	82	107	114	51
4132 S	51	67	81	93	42
4168 B	55	66	81	92	37

Table E6. Pig weight gain pen 21 normal waterer (N)

Pig	Body weight in lb				Weight gain lb
	5/1/71	19/1/71	2/2/71	10/2/71	
2606 B	67	94	121	132	65
2943 S	48	70	83	94	46
2973 B	62	84	107	118	56
2947 B	60	83	108	124	64
2955 B	61	88	100	105	44
2970 S	55	78	94	104	49
2957 B	57	78	93	93	36
2928 B	71	100	102	124	53
2634 B	69	96	123	137	68
2971 S	46	70	82	91	45

Table E7. Feed weight record

Pen	Weight of feed in lb							
	5/1/71	8/1/71	11/1/71	12/1/71	14/1/71	15/1/71	18/1/71	21/1/71
16(G)	200	50	51	50	75	100	75	100
17(N)	202	50	51	100	100	100	200	100
18(G)	200	50	51	50	75	100	75	100
19(N)	200	50	51	50	100	100	75	100
20(G)	200	50	51	50	100	100	75	100
21	200	50	51	50	100	100	100	200

Table E7. (Continued)

Pen	Weight of feed in lb							
	22/1/71	27/1/71	29/1/71	1/2/71	5/2/71	8/2/71	Remnant	Total
16(G)	100	100	0	150	200	100	110	1,241
17(N)	100	100	100	200	200	200	71	1,732
18(G)	100	100	100	150	200	100	62	1,389
19(N)	100	100	100	200	200	100	64	1,462
20(G)	100	100	100	150	200	50	68	1,358
21(N)	100	100	0	150	200	100	76	1,425

Table E8. Pen weight gain totals

	Treatment		
	G	N	
Block	401	616	1017
	482	550	1032
	430	520	956
	1313	1692	

$\Sigma Y = 3005$ $\Sigma Y^2 = 159,515$ Myy = 150,500.4

Table E9. Weight gain ANOV (completely randomized design)

Tyy = 2394 Eyy = 771.3 Syy = 5849.3

Source of variation	Degrees of freedom	Sum of squares	Mean square	F statistic
Mean	1	150,500.4	150,500.4	
Treatments	1	2,394	2,394	12.42
Experimental error	4	771.3	192.8	($F_{1,4}$=7.71 at 5%)
Sampling error	54	5,849.3	108.3	
Total	60	159,515.0		

Table E10. Weight gain ANOV (randomized complete block design)

S_{ft} = 3165.3 Syy = 5849.3 Byy = 162.1

Tyy = 2394 Eyy = 609.2

Source of variation	Degrees of freedom	Sum of squares	Mean square	F statistic
Mean	1	150,500.4	150,500.4	
Blocks	2	162.1	81.1	
Treatments	1	2,394	2,394	7.86
Experimental error	2	609.2	304.6	($F_{1,2}$=18.5 at 5%)
Sampling error	54	5,849.3	108.3	
Total	60	159,515.0		

Table E11. Feed consumed totals

	Treatment G	Treatment N	
Block	1,261	1,732	2,973
	1,389	1,462	2,851
	1,358	1,425	2,783
	3,988	4,616	

$\Sigma Y = 8607$ $\Sigma Y^2 = 12,481,459$ $M_{yy} = 12,346,741.5$

Table E12. Feed consumed ANOV (completely randomized design)

$T_{yy} = 57,125.2$ $E_{yy} = 77,592.3$

Source of variation	Degrees of freedom	Sum of squares	Mean square	F Statistic
Mean	1	12,346,741.6	12,346,741.6	
Treatments	1	57,125.2	57,125.2	2.94
Experimental error	4	77,592.3	19,398.1	($F_{1,4}$ = 7.71 at 5%)
Total	6	12,481,459.1		

Table E13. Feed consumed ANOV (randomized complete block design)

$B_{yy} = 9268$ $T_{yy} = 57,125.2$ $E_{yy} = 68,324.3$

Source of variation	Degrees of freedom	Sum of squares	Mean square	F statistic
Mean	1	12,346,741.5	12,346,741.5	
Blocks	2	9,268	4,634	
Treatments	1	57,125.2	57,125.2	1.67
Experimental error	2	68,324.3	34,162.2	($F_{1,2}$=18.5 at 5%)
Total	6	12,481,459.1		

Table E14. Feed conversion

	Treatment		
	G	N	
Block	3.09	2.81	5.90
	2.88	2.66	5.54
	3.16	2.71	5.87
	9.13	8.18	

$\Sigma Y = 17.31$ $\Sigma Y^2 = 50.1439$ $M_{yy} = 49.9394$

Table E15. Feed conversion ANOV (randomized complete block design)

$B_{yy} = 0.0399$ $T_{yy} = 0.1504$ $E_{yy} = 0.0142$

Source of variation	Degrees of freedom	Sum of squares	Mean square	F statistic
Mean	1	49.9394	49.9394	
Blocks	2	0.0399	0.0199	
Treatments	1	0.1504	0.1504	21.2
Experimental error	2	0.0142	0.0071	($F_{1,2}$ = 18.5 at 5%)
Total		50.1439		